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Ecological Factors Controlling Biodiversity in the British Countryside (ECOFACT)

MODULE 6 - The Causes of Changes in Biodiversity

Interim Report

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EcoFact Module 6 - Timetable

Activity	1996					1997										1998						
	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M
A. Upland studies																						
A1. Collect data/samples by field survey																						
A2. Process samples and complete chemical analysis																						
A3. Carry out statistical analysis of vegetation																						
A4. Correlate botanical data with pollution loadings																						
A5. Review available models																						
B. Impacts Table																						
B1. Develop methods																						
B2. Test trial impacts table through use of case studies																						
B3. Input final data to impacts table																						
B4. Write report																						
C. Verge study																						
C1. Collect data by field survey																						
C2. Carry out preliminary botanical analysis																						
C3. Liaise with Cumbria CC																						
C4. Design and circulate questionnaire																						
C5. Input data and carry out final botanical analysis																						
D. River Corridors																						
D1. Develop methods and carry out field survey																						
D2. Discuss database management with IFE																						
D3. Analyse management data																						
E. Hedge transect																						
E1. Develop methods and carry out field survey																						
E2. Analyse botanical data																						
F. Impact of changes in woodland canopy																						
F1. Analyse conifer woodland species closure in upland woods																						
F2. Analyse woodland cover in lowland woods																						
G. Species cover and Ellenberg scores																						
G1. Correlate species cover change and Ellenberg scores																						

EcoFact Module 6 - Timetable

[illegible]

Introduction

This document contains all the main analyses due to be completed in the project with the exception of the river bank analysis. Initial interpretations have been provided as a basis for discussion at the TSG meeting. The rest of the project will involve further interpretation of the results and synthesis into an integrated document.

The work on impacts tables being carried out by Roy Haines-Young is continuing and two small sub-contracts on the interpretation of appropriate management practices to Carys Swanwick at the University of Sheffield and FWAG will be arranged after this meeting.

INFLUENCE OF GRAZING AND POLLUTION ON UPLAND VEGETATION

Vegetation data were analysed in presence/absence form.

Independent variables used in the analysis were:

- Northing, Easting (geographical location)
- Altitude of the plot (location)
- Nitrogen deposition, as extracted from critical loads atmospheric deposition maps
- Nitrogen (% of dw) sampled in *Calluna*, *Sphagnum* and *Rhacomitrium* samples in square
- Volume of dung collected in square
- Number of grazed shoots: a measure of grazing intensity
- Soiltype, from the Countryside Survey data
- Soil pH, as measured from soil samples taken in the squares

Partially dependent variables were calculated for each plot from the species data and the recalculated Ellenberg scores and CSR scores. The following partially dependent variables were used in the analysis alongside the independent variables:

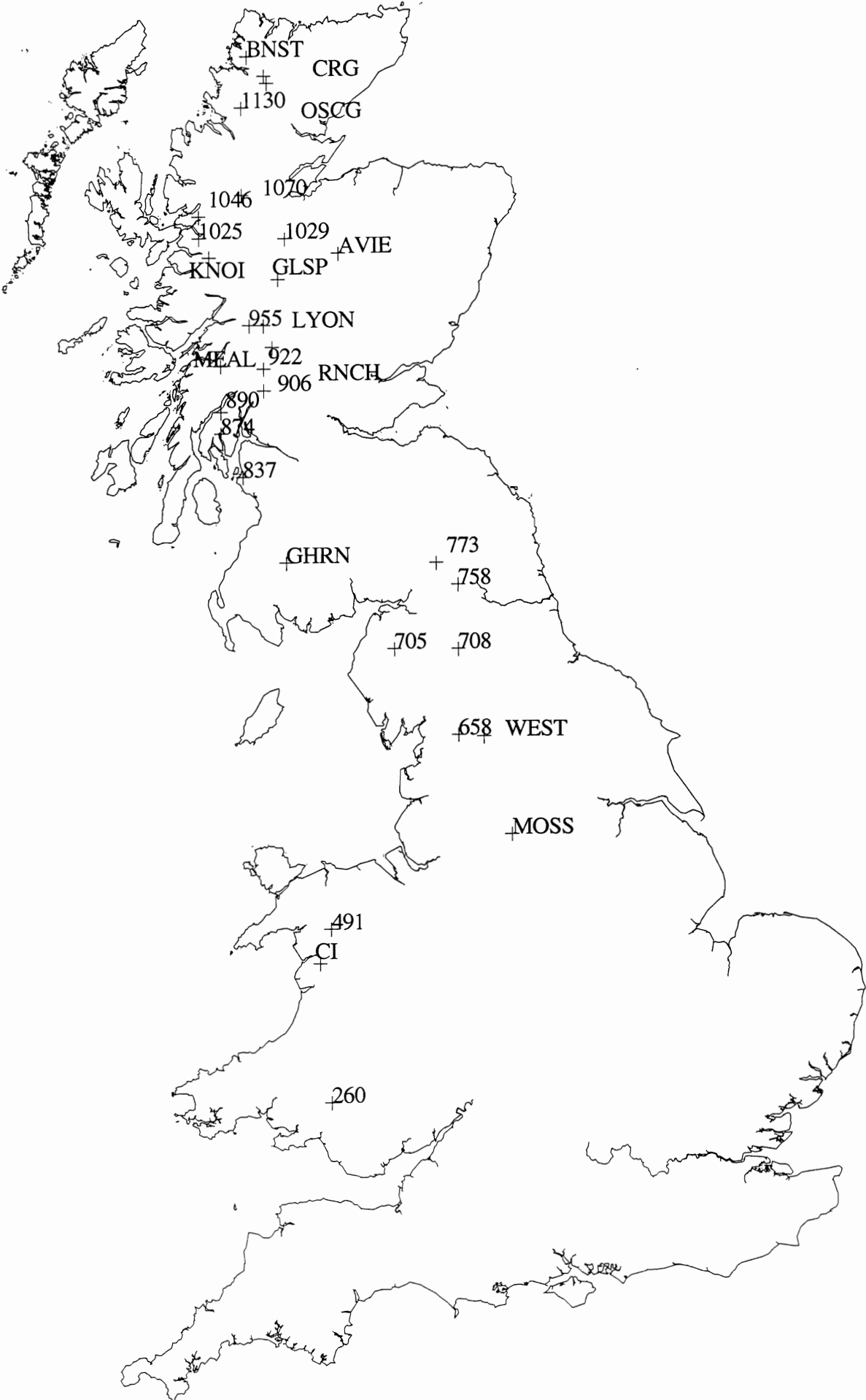
- Mean Ellenberg moisture (F), light (L), pH (R), nutrients (N), temperature (T), continentality (K) scores
- Mean C, S and R scores.

Vegetation was compared and allocated to a vegetation plotclass from the Countryside Survey 1990. Environmental and partially dependent variables were initially analysed using PCA. The whole dataset in its entirety was then analysed in CANOCO, using CCA. The data were analysed as a set, because the interest in this specific case was in the influence and interactions of variables on each other. As grazing is carried out on vegetation, these two are inextricably mixed and cannot be analysed separately.

RESULTS

The result of the PCA on the independent environmental variables and partially dependent species variables is shown in figure 1. This shows a gradient through the data, with at one end high Ellenberg temperature and pH values and predominantly ruderal growing strategies. At the other extreme, growing conditions are suited to stress tolerators with high moisture and light values (open vegetation). In between these two extremes, conditions with high continentality, high atmospheric Ndeposition and high % tissue N suit competitive growing strategies. These sites are found more in the East, which in this survey are the more continental sites (see map).

Map 1 Distribution of sites for 1996 upland grazing and pollution study



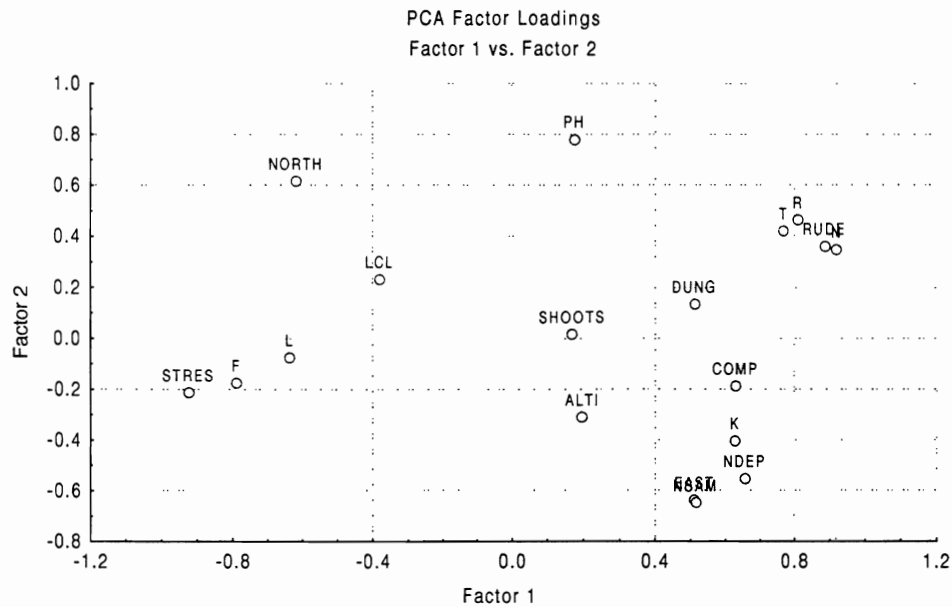


Figure 1. Result of the PCA, showing the major gradients in the independent environmental variables and the partially dependent variables. Alt = altitude, Comp = competitors in vegetation; East = Easting; F = Ellenberg moisture indicator score; K = Ellenberg continentality indicator score; L = Ellenberg light indicator score; Lcl = landclass; Ndep = Atmospheric Nitrogen deposition; North = Northing; Nsam = % Nitrogen in plant tissue; PH = Soil pH; Rude = ruderals in vegetation; Shoots; number of grazed shoots; Stres = stress tolerators in vegetation.

Figures 2 to 4 show results of the CANOCO analysis on vegetation and environmental data using only the independent variables. Arrows show the significant variables found in the analysis. The length of the arrow is related to the variance explained by the variable, the proximity to an axis is a measure for how related the variable is to the CANOCO factors. Hence, in figure 2, the most significant amount of variation is explained by the North-gradient ($p < 0.01$), followed by the dung volume ($p < 0.01$), then the soil pH ($p < 0.01$), altitude ($p < 0.01$), Easting ($p < 0.01$), and the nitrogen deposition ($p < 0.04$). The first CANOCO axis is closely related to dungvolume and Nitrogen deposition, the second to soil pH. The most important variable, Northing, is related to both axes.

The sample sites in figure 2 are therefore distributed according to their geographical position; with north Scottish sites to the left and English and Welsh sites to the right. There is an obvious inverse correlation between the North gradient and both atmospheric N deposition and dung volume found in a square.

Figure 3 shows the same result as figure 2, but for each sample site the aggregate vegetation class (from CS1990) is depicted. It is clearly visible in this figure how dry heath and bog vegetation (classes 8 and 9) are predominantly present in the more atrophic, acid, Northern sites, whereas the eutrophic Welsh and English sites, with higher Ndeposition and dung volumes and higher soil pH, are more moorland grasslands (class 7).

Figure 4 shows the same result as figure 2, but here the landclass (from CS1990) is depicted for each sample site. The higher landclasses are all scattered high along the

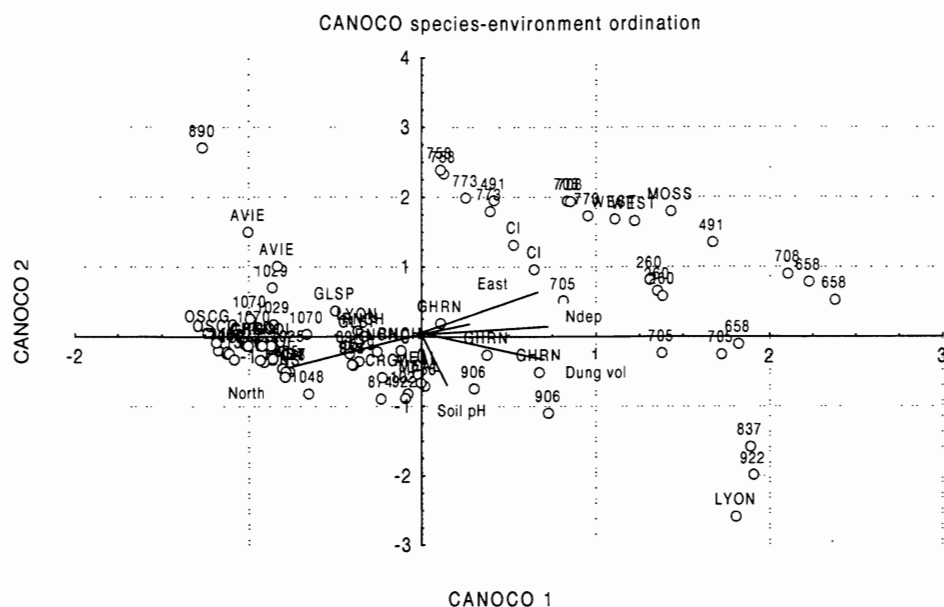


Figure 2: Result of CANOCO analysis of principal gradients using independent environmental variables. Numbers above dots represent the CS square number. Arrows depict variables: North = Northing; East = Easting, Ndep = atmospheric N deposition; the arrow between East and Ndep is altitude; Dungvol = dungvolume; Soil pH = Soil pH

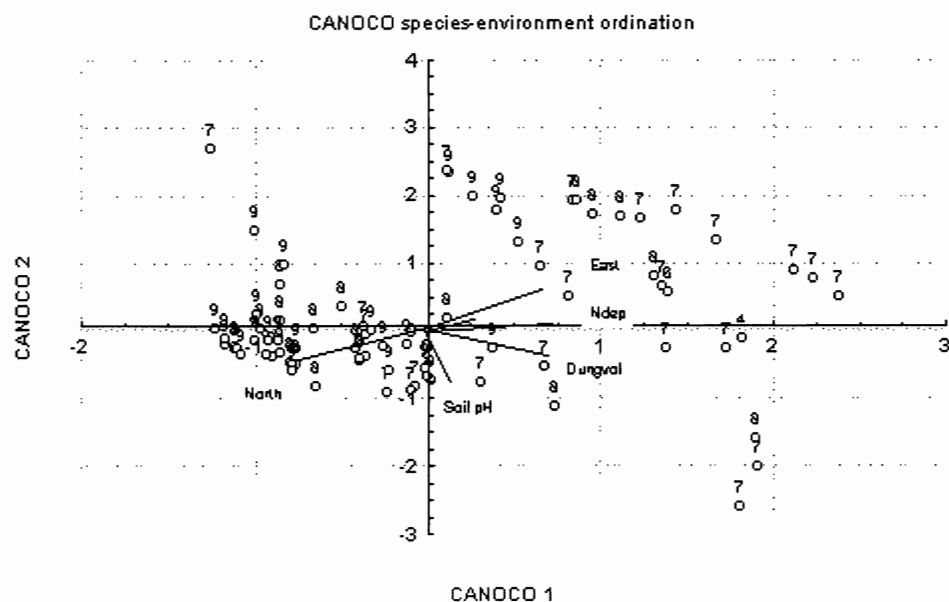


Figure 3: Result of CANOCO analysis of principal gradients using independent environmental variables. Numbers above dots depict classes derived from the Aggregate Vegetation Classes from CS1990. 7 = Moorland/grassland; 8 = dry heathland; 9 = bogs. Arrows depict variables: North = Northing; East = Easting, Ndep = atmospheric N deposition; the arrow between East and Ndep is altitude; Dungvol = dungvolume; Soil pH = Soil pH

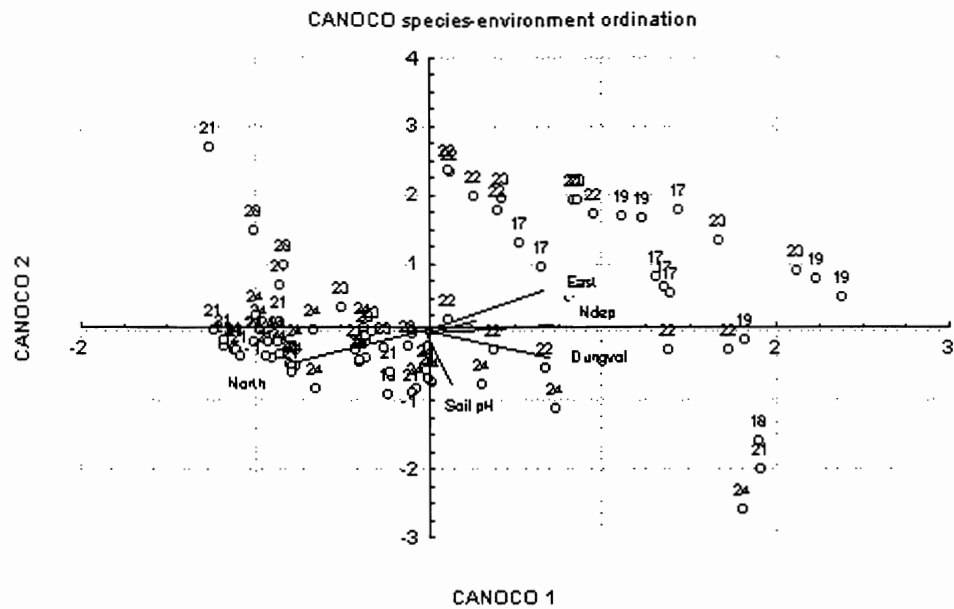


Figure 4: Result of CANOCO analysis of principal gradients using independent environmental variables. Numbers above dots depict landclasses from CS1990. Arrows depict variables: North = Northing; East = Easting, Ndep = atmospheric N deposition; the arrow between East and Ndep is altitude; Dungvol = dungvolume; Soil pH = Soil pH

North gradient, indicating the more extensive landuse forms such as extensive grazing on rougher pastures and rocky uplands that is performed in Scotland on atrophic, acidic soils. More eutrophic sites further south are generally marginal uplands under more intensive grazing regimes.

Figures 5, 6 and 7 show the result of the analysis of the vegetation data with all the environmental variables, both independent and partially dependent. There was a shift in the significance of the variables, compared to the first CCA analysis described above, as a newly introduced variable might explain some additional variation in the data, and make another variable partly redundant. In this case, the North gradient present in the first ordination (figures 2,3,4) is replaced by a score for temperature, which can be assumed to be the more direct effect that latitude will have on growing conditions for vegetation. The latitudinal effect is fortified by the altitudinal effect; hence creating a gradient more or less diagonal through both axis of the scatterplot over which growing conditions become increasingly harsh as temperatures drop at higher altitudes in the North.

The main variables in figure 5 are formed by Ruderal score of the vegetation ($p < 0.01$), Easting ($p < 0.01$), Ellenberg temperature score ($p < 0.02$), Competitor score of the vegetation ($p < 0.01$), altitude ($p < 0.01$), Ellenberg continentality score ($p < 0.03$), dung volume ($p < 0.01$), and nitrogen deposition ($p < 0.04$). Overall ordination was significant at 1%.

The main gradient in the ordination of figure 5 is the Ruderal score of the vegetation, going nearly parallel to the first Canonical axis. It becomes clear from this figure that vegetation with high atmospheric N deposition and high grazing (high dung volumes), and, less importantly, increasing continentality, are characterised by ruderals. In figure 5, Scottish sites remain scattered to the left of the plot which are characterized by low N deposition, low dung volume, low ruderal scores and occur under the harsher growing conditions with low temperatures at higher altitudes.

Figures 6 and 7 show the vegetation classes and the landclasses from CS1990 next to each sample plot. Dry heaths and bogs (Aggregate classes 8 and 9) have low ruderal scores and lower temperatures than the eutrophic moorland grasslands more to the East. These gradients are also visible in the landclasses in figure 7; the higher landclasses (more extensive grazing, rocky landscapes) are scattered where growing conditions are harsh, Nitrogen inputs in the system are low both through deposition and dung.

DISCUSSION AND CONCLUSIONS

Nitrogen inputs in the system are in the form of dung and atmospheric deposition. The high correlations between Nitrogen found in plant tissue on the site and estimated atmospheric deposition (and less importantly also dung volume) make it improbable that there are many other important Nitrogen sources for these upland sites.

There was a significant relationship with the gradient of growing conditions in the data. This was either characterised by Northing or by Ellenberg temperature indicator

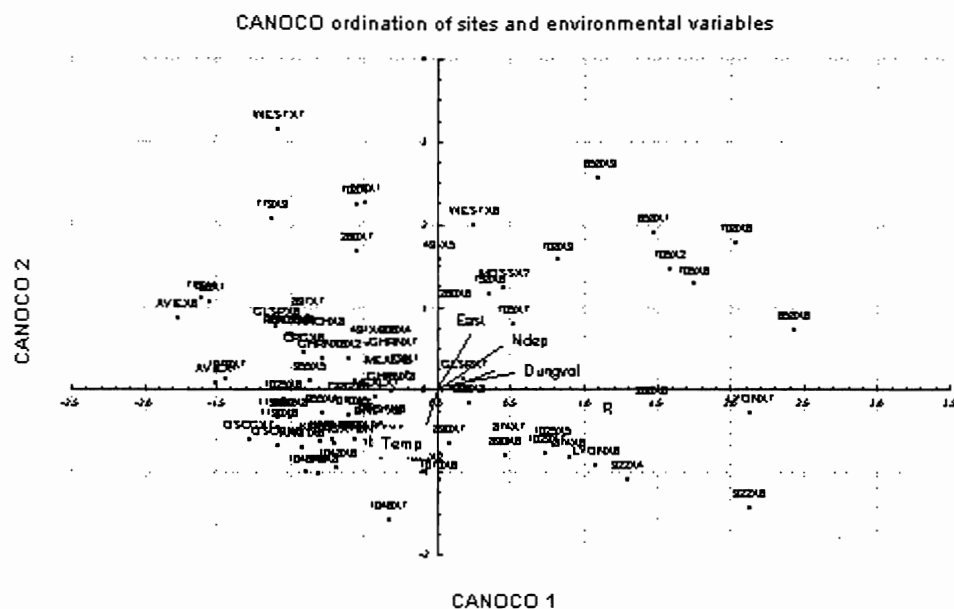


Figure 5: Result of the second CANOCO analysis of principal gradients using independent environmental variables and partially dependent vegetation variables. Numbers above dots depict CS1990 squares. Arrows depict variables: R = ruderal score and K = continentality lie closely parallel to the first axis. Temp = temperature; East = Easting, Ndep = atmospheric N deposition; Dungvol = dungvolume; the arrow between Ndep and dungvolume is altitude.

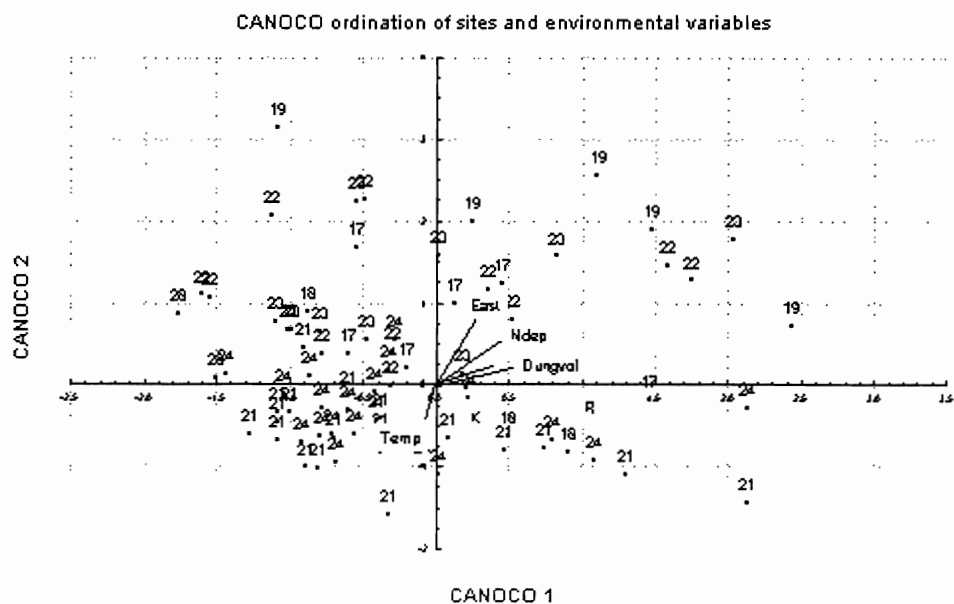


Figure 6: Result of the second CANOCO analysis of principal gradients using independent environmental variables and partially dependent vegetation variables. Numbers above dots depict classes derived from CS1990 Aggregate Vegetation classes: 7 = Moorland/grassland; 8 = dry heathland; 9 = bogs. Arrows depict variables: R = ruderal score and K = continentality lie closely parallel to the first axis. Temp = temperature; East = Easting, Ndep = atmospheric N deposition; Dungvol = dungvolume; the arrow between Ndep and dungvolume is altitude.

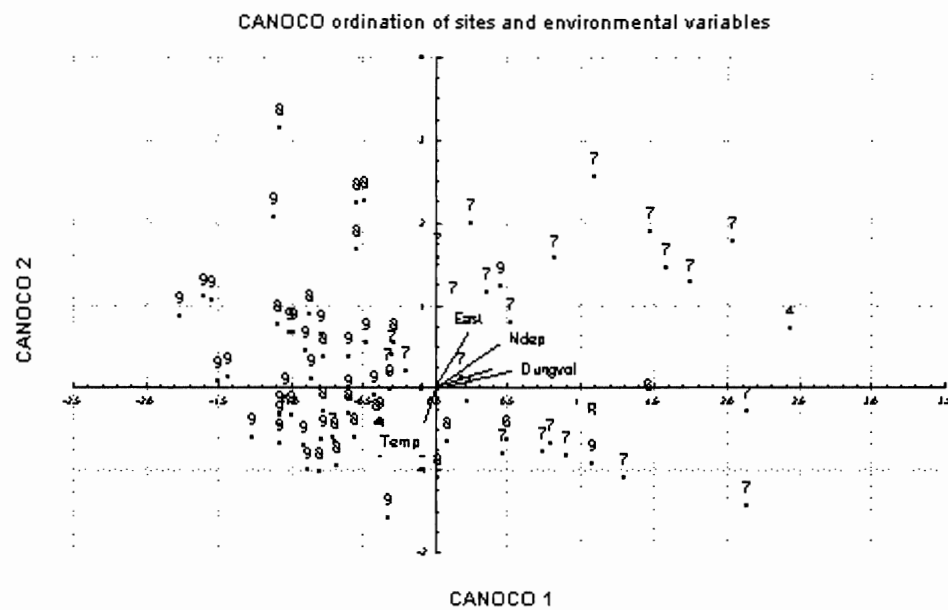


Figure 7: Result of the second CANOCO analysis of principal gradients using independent environmental variables and partially dependent vegetation variables. Numbers above dots depict landclasses from CS1990. Arrows depict variables: R = ruderal score and K = continentality lie closely parallel to the first axis. Temp = temperature; East = Easting, Ndep = atmospheric N deposition; Dungvol = dungvolume; the arrow between Ndep and dungvolume is altitude.

scores, Easting , continentality and altitude, but may also be summarised as upland to lowland, which in turn is expressed by the landclass.

In GB; Nitrogen deposition decreases to the North, and increases to the East, within the range of sites sampled.

The more eutrophic sites were characterised by Aggregate Class 7 (grass mosaic/ moorland vegetation), whereas the atrophic, acidic sites with little N coming in from whatever source were predominantly heaths and bogs. There is a positive feedback mechanism in action because grazing animals all favour more palatable grasses leading to more dung deposition. The present study has established for the first time that there is also a coincidence with deposition because the north of Scotland is dominated by bogs and has low deposition. There was no clear difference between heaths and bogs along the gradients used in this study.

Because of the close relationship between atmospheric N deposition and grazing through N influx in the system, it is difficult to separate their effects on vegetation. It has become clear, however, that these two influences mostly occur simultaneously in the first place, and effects could well be synergistic, as suggested by literature.

Research has suggested that both high atmospheric N deposition and grazing increase the grass cover in the vegetation. The correlative nature of this study does not allow this relationship to be statistically established. However, trends in our data suggest decreasing numbers of heath and bog vegetations and increasing grasslands along the axis of Nitrogen deposition and grazing. While it could be argued that grazing would be biased towards grasslands, this is not true for atmospheric Nitrogen deposition.

Generally, this study has led to the establishment of correlations found in the actual growing environment. More experimental setups following conclusions of this study could lead to establish those relationships.

Impacts Table

Discussion have been held with Dr Roy Haines-Young concerning the structure of the impacts table. Thirty two entries have been constructed and sent for incorporation in the tables, as specified in the timetable. These constitute the most important entries to which further information will be added during December and January.

Changes on Cumbrian road verges between 1992 and 1997

Introduction

Analysis of Countryside Survey data reported in the CS90 main report and as part of EcoFact showed that important changes in botanical diversity and quality had occurred on road verge plots across GB between 1978 and 1990. The importance of road verges as refugia for species associated with unimproved grassland was also highlighted. In an attempt to further elucidate the ecological details of these changes and therefore to generate hypotheses concerning the most likely causes a case study of change in Cumbrian road verges was set up with the following objectives:

- 1) To describe and quantify vegetation change in terms of botanical quality and functional characteristics.
- 2) To explain the greatest amount of floristic variation in time with the minimum of functional characteristics.
- 3) To interpret the model of change in such a way as to hypothesise the most likely factors responsible for shifts in species composition.
- 4) To estimate the importance of positive management and floristic starting point on vegetation change.

Methods

Survey

Survey methods have been summarised in detail elsewhere. In brief the botanical species composition of fixed and relocatable plots was recorded in Summer 1992, Spring 1996, Summer 1996 and Summer 1997. The 1992 survey was originally carried out by the county council as part of a Cumbria wide survey of the road verge resource during which the most species rich grassland sections were identified and quadrat sampled. Many of these were designated as 'special' verges reflecting their very high botanical value. All Cumbrian verges were classified according to vegetation type and each was allocated a specific management regime designed to favour maintenance and expansion of species rich plant assemblages. Each verge management type (VMT) thus encompasses a particular vegetation type associated with which is a cutting regime which in essence prescribes a late first cut for the highly regarded herb rich meadow assemblages (tables 1 & 2). In Spring 1996 thirty verges were randomly selected within which thirty two plots that had been first recorded in 1992, were relocated and re-recorded. Plots measured 1x10m along the road edge. Botanical recording was carried out again in Summer 1996 and 1997.

Analysis

The data set covers five VMTs and both 'special' and non-'special' verge lengths (table 1). Since only presence and absence of species was recorded in 1992 these data are analysed with p/a data for the later years. The analyses require a

repeated measures approach in recognition of the dependence in time between records since they originate from the same fixed location.

Table 1. Data set for analysis.

Designation	Verge Management Type				
	2	4	9	10	11
Special	18	2	2	4	1
Non-special	0	1	4	0	0
Total replicates	18	3	6	4	1

Table 2. Sampled VMTs and their cutting regimes.

Management System Code	Verge Type	First Cut: Safety Swathe Visibility Splays	Full Width of Verge
2	Short Grassland (Spring / Summer Flowering)	15 August - 31 October	15 August - 31 October
4	Grassland with Heath or Woodland to rear or Short Grassland privately cut at rear in Autumn	15 August - 31 October	No cut
9	Species-rich grassland with spring - late Summer flowers	15 September - 30 September	15 September - 30 September
10	Species - rich grassland with Spring - Autumn flowers	1 October - 31 October	1 October - 31 October
11	Coarse Grassland with late Summer flowers	15 May - 31 May	1 October - 31 October

To assess change in quality and diversity between years repeated measures MANOVA tests were carried out on two sets of derived scores for each plot (table 3). Firstly, to assess floristic trends in botanical quality, changes in similarity between verge species composition and target vegetation types were computed: Jaccard qualitative similarity coefficients were calculated between the species composition of each plot at all points in time and the frequency tables of nine sub-communities of the National Vegetation Classification which describe the floristic profile of a) scarce, unimproved calcareous grasslands associated with the limestone plateaux of the Pennines and south Cumbria, and b) uncommon unimproved hay meadow communities. Only species present at a constancy of 3 or more in each NVC unit were used in the matching process. Similarity coefficients range from 0 (total dissimilarity) to 1.

The hump-backed model of Grime (1979) predicts that maximal species richness is associated with intermediate biomass whereas at high standing crop typically high productivity is associated with a less species rich sward dominated by

competitive species adapted to capitalise upon high levels of resource availability. Low species diversity is also likely to be associated with very low biomass where its accumulation is constrained by stress due to nutrient or/and moisture shortages or frequent disturbance, conditions to which only a limited number of species are tolerant. In a second test we therefore explore changes in the proportions of species within each plot that are expected to be associated with either high productivity and minimal disturbance (Competitors including shrubs and trees, Competitive monocots eg. grasses but excluding woody species) or high stress and disturbance (Stress-tolerators and Ruderal species - SR). Changes in their abundance will enable an interpretation in terms of shifts in conditions favourable to different types and numbers of species and linked to this we therefore also analyse simple changes in species count within each plot over time.

For all MANOVA tests, plot scores are proportions and so were arcsin transformed.

Table 3. NVC units against which verge plots were matched to derive similarity scores.

CG10a	<i>Festuca ovina</i> - <i>Agrostis capillaris</i> - <i>Thymus praecox</i> grassland, <i>Trifolium repens</i> - <i>Luzula campestris</i> sub-community.
CG9b	<i>Sesleria albicans</i> - <i>Galium sternerii</i> grassland, Typical sub-community
CG9c	<i>Carex pulicaris</i> - <i>Carex panicea</i> sub-community
MG3a	<i>Anthoxanthum odoratum</i> - <i>Geranium sylvaticum</i> grassland, <i>Bromus hordeaceus</i> sub-community
MG3b	<i>Briza media</i> sub-community
MG5a	<i>Centaurea nigra</i> - <i>Cynosurus cristatus</i> grassland, <i>Lathyrus pratensis</i> sub-community
MG5b	<i>Galium verum</i> sub-community
MG5c	<i>Danthonia decumbens</i> sub-community

The above analyses involve variables selected with a prior expectation of their worth in interpreting change. These were complemented by a multivariate direct gradient analysis using CANOCO. This was carried out so that the influence of, and relationship between species over time could be explored and these relationships explained by the best combination of species attributes even if these are more difficult to interpret. Again we chose a number of variables related to aspects of established strategy (*sensu* Grime, 1979) but also incorporated traits relating to regenerative strategy, seedbank type, maximum canopy height and Ellenberg scores for light, soil reaction and fertility. These variables were chosen since changing fertility and the disturbance effects of management were most likely to be implicated in any changes detected. The combination of variables with the most explanatory power were selected by Monte-Carlo permutation tests.

Because we were interested in common trends between verges and not effects related to the fact that a plot was on a particular verge we used verge ID as a covariable. This was also implemented for the MANOVA tests.

McNemar χ^2 tests were used to detect significant changes in species frequency between years and are used in conjunction with PCCA output to interpret change.

Separate analyses of 1992 data with the 1996 records were carried out to corroborate changes detected over the longest interval between 1992 and Summer 1997.

Results

Changes in botanical quality

Similarity with six out of eight NVC units declined between 1992 and 1997 (table 4 and figure 1). The same trends occurred across VMT units with no significant interaction for any coefficient detected between two groups; one comprising VMT 2 the other comprising plots in VMTs 4, 9, 10 and 11.

Analyses for 1992 to Spring and Summer 1996 data yield the same significant effect of time overall. However in these analyses significant reductions in similarity coefficients were only detected for CG9b, MG5b and MG5c although trends for all units were downward. Note that the highest similarity coefficients in both VMT groups are for the unimproved neutral grassland community MG5.

Figure 1. Change in Jaccard similarity coefficient between 1992-97. Interaction between VMT over time ns; $p > 0.1$. Only sig changes in arcsin transformed values shown.

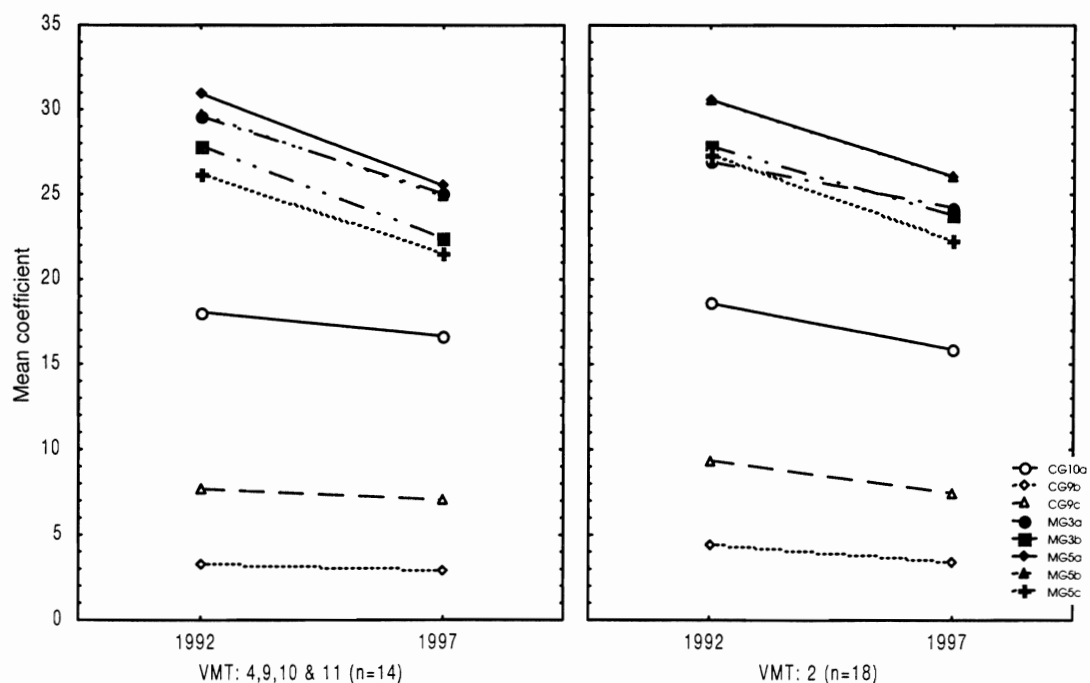


Figure 2 and table 4 show that between 1992 and 1997 species richness declined significantly along with the proportional contribution of stress-tolerant and ruderal species per plot whilst there was an increase in the mean proportion of

competitive species in each plot (note that the two groups do not include all species so that neither score is redundant). However examination of the absolute values indicates that SR species still make up the largest proportion of the species present in the sample data in both years.

Analysis of changes in the same variables between 1992 and 1996 verify the 92 to 97 trends.

Figure 2. Change in mean values of proportional contributions of summary CSR groups and species richness.

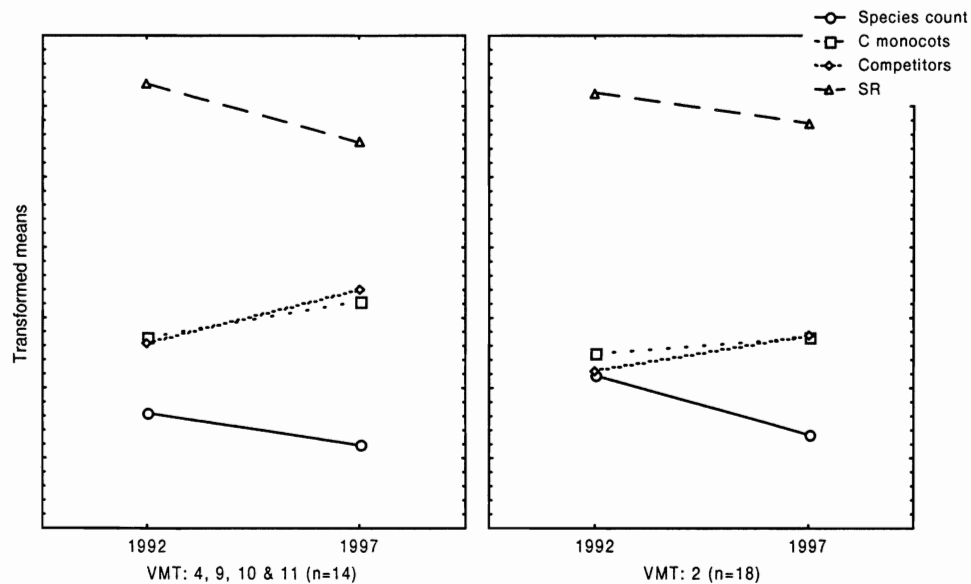


Table 4. MANOVA results for change in similarity coefficients and plot variables between 1992 and 1997.

	F	p		F	p
CG10a	8.99	**	Species richness	5.95	*
CG9b	0.57	ns	C monocots	3.86	ns
CG9c	1.75	ns	Competitors	8.04	**
MG3a	16.3	***	SR species	5.14	*
MG3b	25.54	***			
MG5a	34.04	***			
MG5b	30.29	***			
MG5c	39.30	***			

Multivariate direct gradient analysis

As shown in table 5 eigen values are low for all axes including 1 and 2 which usually pick out the major gradients in the species data. After fitting covariables (verge location) which explain 33.9% of the total variation, only 6.6% of the remaining variation is explained by the best fitting attributes of persistent seedbank, seasonal regeneration and vegetative regeneration (figure 3). Axis 1 sample scores appear to be related to change through time (confirmed by figure 4) in turn associated with a shift towards species with a persistent seedbank. Axis 2 is correlated with changes in the representation of species exhibiting seasonal regeneration in gaps and although there appears to be a trend in axis 2 scores over time there was no significant difference between years.

Clearly most of the variance is unexplained. However some evidence of the fact that we have not failed to pick out other gradients comes from an initial DCA carried out to aid response model selection and to check for outliers and passive species by environment correlation. These results showed that axes 1 and 2, unconstrained by any environmental variables, together explain only 11% of the variation. High correlation coefficients between species and environment axes and the best fitted variables indicate the existence of the relationship between attribute and axes although these relationship are very weakly expressed because of noise in the data.

Table 5. Summary of PCCA results for verge data (1992 - 97) constrained by an optimum combination of 3 attribute scores for plots.

Axes	1	2	3	4	Total inertia
Eigenvalues	.118	.111	.088	.081	5.116
Species-environment correlations	.897	.901	.865	.835	
Cumulative percentage variance					
of species data	3.5	6.8	9.4	11.8	
of species-environment relation:	29.6	57.5	79.7	100.0	
Sum of all unconstrained eigenvalues (after fitting covariables)					3.380
Sum of all canonical eigenvalues (after fitting covariables)					.399

Figure 3. Site scores for Cumbrian road verges as constrained by an optimum combination of 3 attributes. PCCA biplot with verge as covariable.

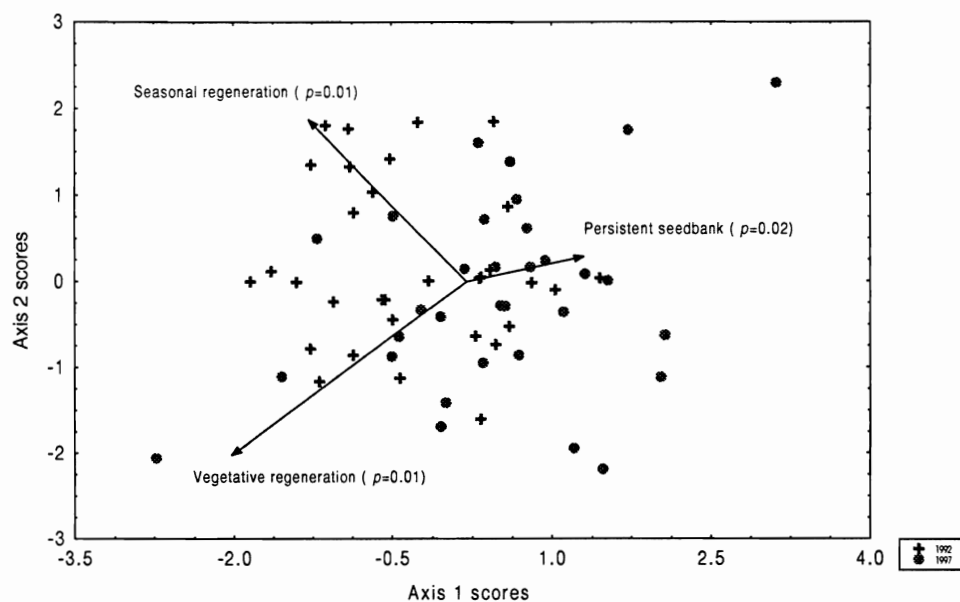
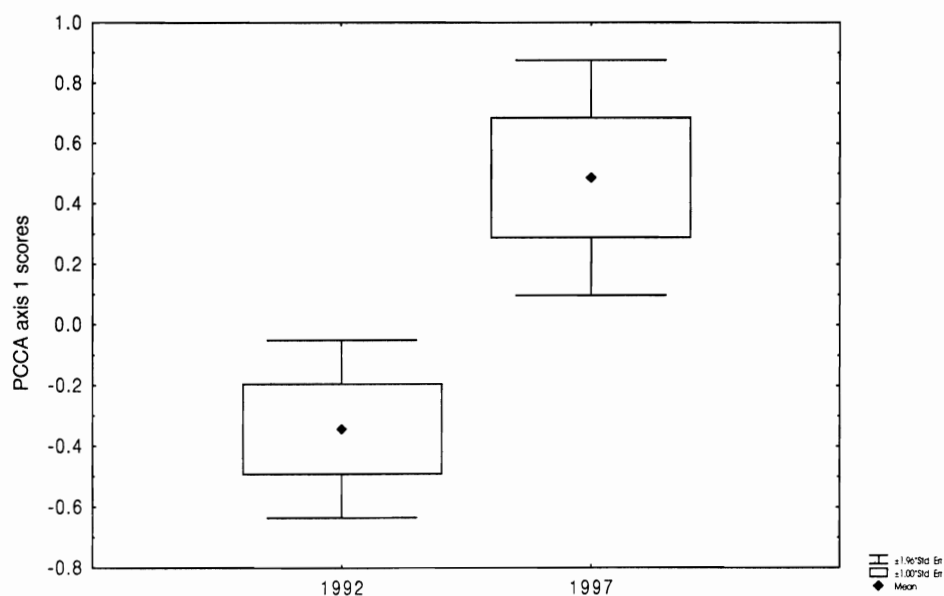


Figure 4. PCCA axis 1 sample scores constrained by best combination of attributes. Difference between years significant ($p=0.013$, Wilcoxon matched pairs).



Examination of trends in individual species (table 6) with different regenerative strategies indicates that increasing species with persistent seedbanks include *Agrostis stolonifera*, *Polygonum aviculare* and *Matricaria matricoides*. All of which have shown a consistent trend from 1992 through 1996 to 1997. Significantly

decreasing species exhibiting seasonal regeneration in vegetation gaps include the unimproved grassland species *Trifolium pratense* and *Briza media* whilst non-significant downward trends in seasonal regenerators are apparent for other species rich verge plants such as *Centaurea nigra*, *Cynosurus cristatus*, *Geum rivale* and *Leucanthemum vulgare*.

Table 6. Significant changes in between-plot frequency in Cumbrian verges between 1992 and 1996-97, n=32, total number of tests = 114.

Species	Sig 92-96	Sig 92-97	Direction	Freq '92	Freq'96	Freq'97
<i>Leontodon hispidus</i>	**	*	down	8	0	1
<i>Plantago lanceolata</i>	ns	*	down	25	19	17
<i>Polygonum aviculare</i>	ns	*	up	3	8	12
<i>Ranunculus acris</i>	ns	*	down	13	10	4
<i>Agrostis stolonifera</i>	ns	**	up	23	29	32
<i>Matricaria matricoides</i>	***	**	up	0	12	11
<i>Poa pratensis</i> agg.	**	**	up	11	23	24
<i>Veronica chamaedrys</i>	ns	**	down	13	10	3
<i>Holcus lanatus</i>	**	***	down	27	16	12
<i>Briza media</i>	*	ns	Down	6	0	3
<i>Trifolium pratense</i>	*	ns	Down	21	12	16
<i>Alopecurus pratensis</i>	**	ns	Up	0	9	4
<i>Vicia cracca</i>	**	ns	Down	8	0	4

Table 7. Trends in plant species exhibiting a seasonal regenerative strategy.

Species	Regenerative strategy							Net Change	Established strategy	Sig change
	V	Bs	S	W	92	96	97			
<i>Achillea millefolium</i>	1	0	1	0	12	10	8	-4	CR/CSR	✓
<i>Briza media</i>	1	0	1	0	6	0	3	-3	S	
<i>Centaurea nigra</i>	1	0	1	0	16	14	9	-7	CSR	
<i>Chaerophyllum temulentum</i>	0	0	1	0	5	0	0	-5	R/CSR	✓
<i>Cruciata laevipes</i>	1	0	1	0	10	5	4	-6	CSR	
<i>Cynosurus cristatus</i>	0	0	1	0	13	11	8	-5	CSR	
<i>Geum rivale</i>	0	0	1	0	12	10	8	-4	S/CSR	
<i>Heracleum sphondylium</i>	0	0	1	0	15	14	9	-6	CR	
<i>Holcus lanatus</i>	1	1	1	0	27	16	12	-15	CSR	
<i>Leucanthemum vulgare</i>	1	1	1	0	4	3	1	-3	C/CSR	
<i>Succisa pratensis</i>	0	0	1	0	3	0	1	-2	S	
<i>Trifolium pratense</i>	0	1	1	0	21	12	16	-5	CSR	
<i>Trisetum flavescens</i>	1	0	1	0	12	9	8	-4	CSR	

Discussion

The variables that explained most of the variation in the verge data refer to aspects of the regenerative strategy of plant species and are not linked to eutrophication effects. Mindful of the small amount of variation explained, our results suggest that increasing fertility resulting from neglect and litter accumulation or from atmospheric or roadside deposition are less important in these data than disturbance effects. However because we did not experimentally manipulate

treatments and had no unmanaged control we cannot partition variation among effects but must interpret results in the light of known management changes as well as making a contrast with GB wide trends. We do know that the positive verge management system was first implemented in 1995 and for all except one plot this would have meant a 1m safety swathe cut between mid-August and mid-October (the whole verge width is also cut once every two years for special verges and every four years for non-specials). Such management is essentially designed to mimic a late hay cut thereby favouring regeneration by seed which is especially important for annuals such as *Vicia sativa* and *Rhinanthus minor*. In the ten years or so before 1995 the 1m safety swathe could be cut at any time of the year depending upon sward growth and receipt of complaints whilst the remainder of the verge might remain uncut for five or more years. Other changes to have occurred in the last twenty years are a substantial increase in average daily traffic at least on A roads in the sample region between 1984 and 1997 (AADT data, Cumbria County Council) and the cessation of the practice of trimming back shallow growth of vegetation over the metalled road edge.

It is unlikely that the effects of the new management system will have been detected within the space of 8 months since the first late cut and vegetation recording in Spring 1996. Even though management experiments on species rich road verges and in Pennine hay meadows elicited treatment effects within four to six years of the start of experiments (Smith & Rushton, 1994, Parr & Way, 1988, Melaman et.al. 1988) vegetation response is likely to differ greatly between verges reflecting large variability in residual fertility, the initial floristics of the vegetation and distance to nearby seed sources (Egler, 1954, Tilman, 1997). For example Willems & van Nieuwstadt (1997) showed that even though fertility and biomass could be reduced within six years in a mown and previously fertilised grassland, the establishment of a species rich sward could take as long as 25 years in the absence of propagules whilst the reduction in competitive herbs in the short term led to a drop in species richness. In our analyses such local effects are likely to influence verge-specific responses and it is unsurprising, given our small sample size, that the covariable explains around 5 times as much variation as the attributes fitted to the comparatively weak cross-plot response.

It is likely that the detected reductions in species richness, stress-tolerators and ruderal species, reductions in similarity with target NVC units and increases in the proportion of competitive species all suggest an effect due to the momentum of infrequent or inappropriately timed disturbance in the past 20 years. These trends parallel changes in many road verges throughout GB where unimproved grassland species declined between 1978 and 1990 whilst the processes at work appeared to be eutrophication and/or dereliction (EcoFact 1A report). As a group, species that exhibit seasonal regeneration include many autumn germinating grasses as well as spring germinating tall herbs and woody plants (Grime, 1979) however between 1992 and 1997 a general downward trend in seasonal regenerators in our data notably encompassed unimproved grassland species such as *Briza media*, *Trifolium pratense*, *Cynosurus cristatus*, *Leucanthemum vulgare*, *Centaurea nigra* and *Geum rivale*. Only the first two showed significant reductions but the coincident drop in a group of taxa with a common regenerative strategy suggests a response to the unavailability of gaps in the sward at appropriate times of the year.

Species that increased in the data would appear to contradict the notion of reduced gap availability since they are associated with regeneration from a persistent buried seedbank. However their increase is probably linked to the heterogeneity that

characterise the road verge plots themselves and the fact that the species concerned are well adapted to take advantage of the unpredictable but often intense disturbance that prevails and is likely to have increased in the 20 to 30 cm strip of trampled, open vegetation at the extreme road edge. In this group significantly increasing plants included *Polygonum aviculare*, *Matricaria matricoides* and *Agrostis stolonifera*. Other increasers characterised by a persistent seedbank included *Poa annua*, *Poa pratensis* agg. and *Plantago major*. Their increase is likely to have been facilitated by the lack of trimming at the edge of the road surface (M.Clowes pers.comm.) and possibly increases in road traffic aiding dispersal and increasing disturbance at the verge edge (Schmidt, 1989).

Conclusions

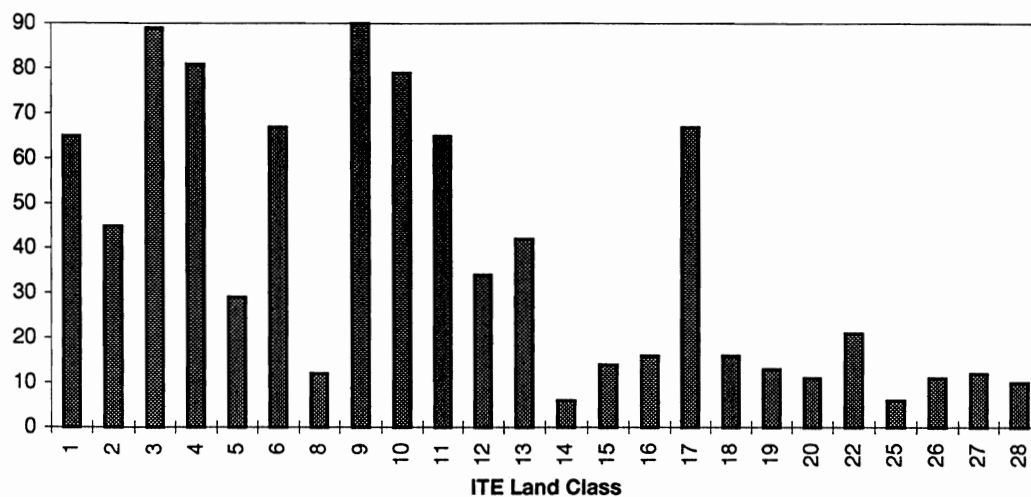
- Longer term monitoring of a larger number of plots plus unmanaged controls is required to capture future changes due to verge management.
- Two trends were detected but both are likely to be occurring in different parts of the plot and for different reasons.
- Firstly a decline in quality was seen between 1992 and 1996/97 with a reduction in unimproved grassland species. This trend has been seen throughout many GB road verges and in the Cumbrian sample is likely to be linked to the ongoing effects of past management. The corollary is that it is too early to detect the effect of the new management system although published evidence on experimental vegetation management would unequivocally favour a regime of the kind that is now in place.
- An increase in diminutive, ruderal plants was detected. This is likely to be a response restricted to the road edge and maybe linked to increased trampling of sparse vegetation in this narrow strip.

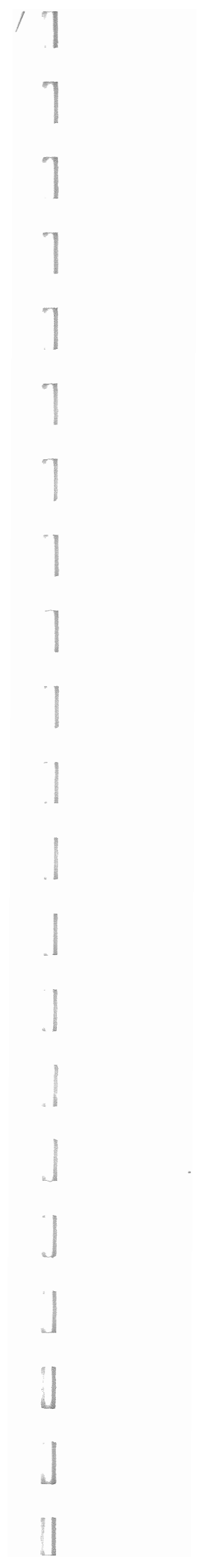
River corridors

Hugh Dawson of IFE is currently analysing the relationship between the vegetation in plots sampled along riversides in 1996. ITE have sent the land class information to enable the analysis to take place.

Figure 1 shows the land class distribution of the total dataset of 901 watercourses sampled for this project. The analysis will proceed by making links between management impacts in these samples and change in the vegetation.

Figure 1. Numbers of watercourse samples by ITE Land Class.





Species distribution within Cumbrian hedgerows

The distribution of plant species within Cumbrian hedgerows and the ecological factors responsible for differences in species composition between hedge types and at a smaller scale within the hedge are examined in this section.

Methods

A stratified, random sample of 22 1km squares from throughout Cumbria provided a framework for the work. In a previous study, the hedgerows were classified on the basis of their species composition into four broad classes hence referred to as Vegetation Classes (VCs).

Within these 1km squares, a total of 162 1m long x 0.5m wide transects were laid from the centre of the hedge, at right angles to the line of the hedge. Each transect was then divided into four 25cm bands, with Band 1 occupying the innermost position of the hedge and Band 4 the outermost. This gave a total of 648 quadrats (162 from each band) from which species presence and cover was recorded. Thus differences in species composition could be investigated in terms of both bands and VCs.

The hedgerow vegetation

Table 1 shows the differences between the bands in terms of species composition. Some species such as *Arrhenatherum elatius* and *Galium aparine* are frequent in all bands whilst others such as *Bellis perennis* are restricted to the outermost band. The innermost band has a higher constancy of shrub species such as *Crataegus monogyna* whilst shrub species have a lower constancy and grassland species such as *Trifolium repens* have a higher constancy in the outermost band

Table 2 shows the differences in species composition between the broader Vegetation Classes, with VC4 having a greater proportion of wetland plants such as *Phragmites australis*.

Figure 1 shows the relative proportions of herb and shrub species in different bands and VCs. For all VCs there is a noticeable decrease in the number of shrub species and increase in herb species from the centre of the hedge. The differences in cover of herb and shrub species are given in Table 3 showing that shrub species have the greatest cover in the innermost band.

Table 3: % cover of herb and shrub species

		VC1	VC2	VC3	VC4
Band 1	Herb	30	25	20	35
	Shrub	65	70	75	65
Band 2	Herb	55	60	55	75
	Shrub	45	40	40	40
Band 3	Herb	90	90	85	100
	Shrub	15	15	15	15
Band 4	Herb	95	100	95	110
	Shrub	5	0	5	10

Multivariate analysis

In order to explore the major vegetation gradients present, the 648 quadrats were subjected to ordination by Detrended Correspondence Analysis (Hill 1979a) using DECORANA and classification by TWINSpan (Hill 1979b). The mean scores for each band on the first two ordination axes together with their standard error are plotted in Figure 2, showing a difference between the scores for each band on the first two ordination axes, with Band 1 having the highest score and Band 4 the lowest on both axis 1 and 2.

TWINSpan analysis was carried out to two levels of division giving 4 groups. Figure 3 shows the composition of the groups in terms of band membership and illustrates that Groups 1 and 2 are dominated by samples from Bands 1 and 2 whilst Groups 3 and 4 are dominated by samples from Bands 3 and 4. The results of the multivariate analysis therefore emphasise the differences in species composition operating on a small scale with distance from the centre of the hedge.

Ellenberg indicator values

Ellenberg indicator values for light, fertility, pH and moisture (as recalibrated by M. O. Hill) were assigned to each species to enable the vegetation to be interpreted in terms of simple environmental gradients. The mean value for each of the above conditions was calculated for each quadrat, excluding those species for which a value was not available. A repeated measures analysis of variance (ANOVA) was carried out in SAS (GLM procedure) to investigate the differences between the bands and VCs, with a separate analysis carried out for each of the above conditions. Results of the ANOVA are given in Table 5 showing the highly significant differences between the VCs and bands in terms of values for light and moisture.

Table 5: Results of repeated measures ANOVA on Ellenberg values for hedgerow quadrats

Ellenberg value	VC	Band	VC x Band
Light	p=<0.001 ***	p=<0.01 **	p=<0.05 *
Moisture	p=<0.001 ***	p=<0.001 ***	p=<0.05 *
Fertility	p=<0.05 *	n.s.	n.s.
pH	n.s.	n.s.	n.s.

Figure 4 illustrates the significant differences in terms of mean Ellenberg values for light between the VCs and how the value increases with distance from the centre of the hedge. This is reflected in the species composition of the bands (Table 1) showing that Band 4, which has the highest value for light, contains the greatest proportion of light-demanding species which tend to be grassland species in contrast to the shade-tolerant shrub species which predominate in the centre of the hedge. An exception to this is VC1, which shows a slight increase followed by a decrease in light value away from the centre of the hedge. This class has the highest initial value for light and so it is possible that any effect of distance from the centre of the hedge may be less pronounced when the hedge is characterised by species with high values for light in the first place.

The differences in Ellenberg values for moisture are shown in Figure 5. VC 4, which contains a greater proportion of wetland species (Table 2), has the highest values for

all the bands, which also increase away from the centre of the hedge. Although the trend is less pronounced for the other VCs than it is for light, there is a significant difference in Ellenberg values for moisture between the bands, with higher mean values in the band furthest away from the centre of the hedge. Whilst this may reflect a difference in conditions of moisture with distance from the hedge, it may also be related to other factors. For example, the band furthest away from the centre of the hedge, which has the highest values for moisture, has a higher proportion of grassland species, some of which may require higher moisture levels such as *Filipendula ulmaria* but may also be more light-demanding.

There was no significant difference between the bands in terms of fertility (Figure 6) at this scale nor pH (Figure 7). However, there is a difference between the VCs in terms of fertility, suggesting that changes in fertility may operate on a larger scale.

The significant differences in light and moisture values between VCs and with distance from the centre of the hedge indicate that these factors operate on both a large and small scale to create heterogeneity in species composition. However several factors may operate simultaneously to influence species composition and it is therefore difficult to partition cause and effect. The fact that a significant difference in species composition occurs over 1m suggests that factors operating at a small scale may have a considerable influence.



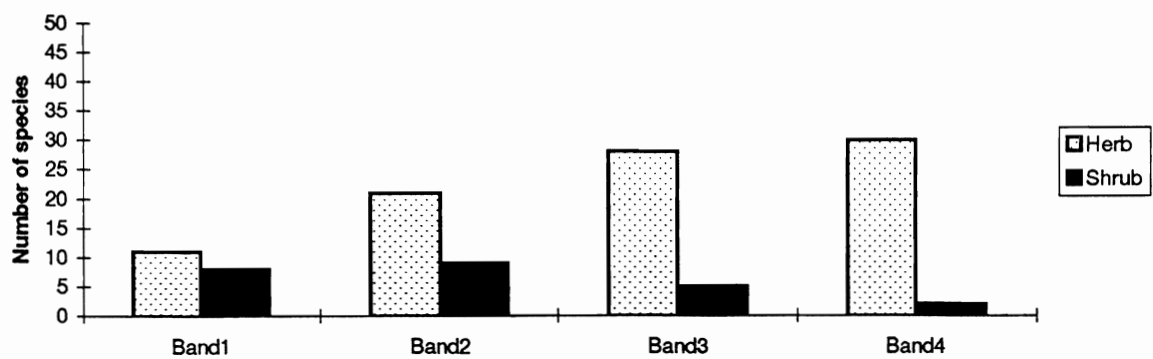
BAND 1		BAND 2		BAND 3		BAND 4	
HERB	% Const	HERB	% Const	HERB	% Const	HERB	% Const
Gali apar	25	Gali apar	40	Arrh elat	50	Arrh elat	35
Arrh elat	20	Arrh elat	35	Loli pere	35	Gali apar	25
Urti dio	20	Urti dio	25	Dact glom	30	Loli pere	25
Fest rubr	15	Dact glom	20	Gali apar	30	Poa triv	25
Poa triv	15	Fest rubr	20	Urti dio	30	Dact glom	25
Dact glom	5	Loli pere	20	Poa triv	20	Urti dio	20
Anth syl	5	Poa triv	20	Anth syl	15	Anth syl	15
Merc per.	5	Ange syl	10	Cera font	15	Fest rubr	15
Desc cesp	5	Geum riva	10	Fest rubr	15	Stel medi	15
		Alop prat	5	Tar: agg	15	Cera font	15
		Cera font	5	Cirs arv	10	Ranu repe	15
		Cirs arv	5	Fili ulma	10	Ranu fica	15
		Merc per	5	Ranu acri	10	Tar: agg	15
		Poa annu	5	Ranu fica	10	Trif rep	10
		Ranu fica	5	Stel medi	10	Cirs arv	10
		Stel medi	5	Vici sati	10	Alop prat	10
		Stell holo	5	Alop prat	5	Vici sati	10
		Tar: agg	5	Desc cesp	5	Fili ulma	10
				Holc lana	5	Holc lana	5
				Merc per.	5	Rume ace	5
				Poa annu	5	Bell per.	5
				Poly vulg	5	Geum riva	5
				Ranu repe	5	Viol rivi	5
				Rume ace	5		
				Sile dioe	5		
				Stach syl	5		
				Stell holo	5		
				Trif rep	5		
				Viol rivi	5		
SHRUB	% const	SHRUB	%Const	SHRUB	% Const	SHRUB	% Const
Crat mon	70	Crat mon	40	Crat mon	10	Rosa	5
Rosa	15	Rubus	15	Prun spin	10	Crat mon	5
Rubus	15	Rosa	10	Rosa	10	Rubus	5
Hede heli	10	Cory avel	5	Rubus	10	Prun spin	5
Prun spin	10	Fagu sylv	5	Hede heli	5		
Ulex euro	10	Hede heli	5				
Cory avel	10	Prun spin	5				
Loni peri	5	Ulex euro	5				
Fagu sylv	5						
Ilex aquil	5						
Samb nigr	5						
Frax exce	5						
Malu syl	5						

Table 1: Differences in species composition between the bands shown as % constancy. (Species with constancy <5% are not shown)

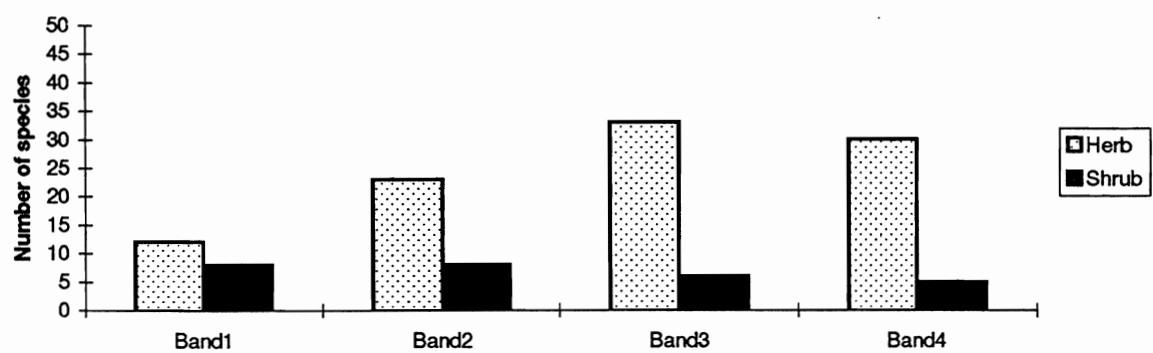
VC1	% Const	VC2	% Const	VC3	% Const	VC4	% Const
HERB							
Lolium perenne	55	Arrhenathrum elatius	45	Arrhenathrum elatius	40	Arrhenathrum elatius	45
Poa trivialis	40	Lolium perenne	40	Galium aparine	30	Urtica dioica	35
Dactylis glomerata	35	Galium aparine	40	Urtica dioica	25	Galium aparine	35
Festuca rubra	35	Urtica dioica	25	Hedera helix	20	Filipendula ulmaria	15
Urtica dioica	20	Dactylis glomerata	20	Dactylis glomerata	20	Deschampsia cespitosa	15
Cerastium fontanum	15	Anthriscus sylvestris	20	Anthriscus sylvestris	15	Anthriscus sylvestris	15
Arrhenathrum elatius	15	Poa trivialis	15	Poa trivialis	15	Poa trivialis	15
Cirsium arvense	15	Taraxacum agg.	10	Festuca rubra	15	Festuca rubra	15
Galium aparine	15	Stellaria media	10	Lolium perenne	10	Vicia sativa	10
Stellaria media	15	Alopecurus pratensis	10	Ranunculus ficaria	10	Geum rivale	10
Ranunculus repens	15	Cerastium fontanum	10	Taraxacum agg.	10	Lolium perenne	5
Taraxacum agg.	15	Mercurialis perennis	10	Alopecurus pratensis	10	Adoxa moschatellina	5
Poa annua	10	Trifolium repens	10	Mercurialis perennis	10	Dactylis glomerata	5
Ranunculus ficaria	10	Vicia sativa	5	Ranunculus repens	5	Viola riviniana	5
Trifolium repens	10	Holcus lanatus	5	Stellaria holostea	5	Galium odoratum	5
Bellis perennis	5	Ranunculus ficaria	5	Cerastium fontanum	5	Phragmites australis	5
Rumex acetosa	5	Alliaria petiolata	5	Dryopteris filix-mas	5	Rumex acetosa	5
Viola riviniana	5	Cirsium arvense	5	Polypodium vulgare	5	Stachys sylvatica	5
Matricaria matricarioides	5	Glechoma hederacea	5	Vicia sativa	5	Cerastium fontanum	5
Conopodium majus	5			Pteridium aquilinum	5	Cirsium arvense	5
Veronica chamaedrys	5			Cirsium arvense	5	Silene dioica	5
Holcus lanatus	5			Rumex acetosa	5	Stellaria holostea	5
Glechoma hederacea	5			Viola riviniana	5	Geranium robertianum	5
Plantago lanceolata	5					Plantago lanceolata	5
Ranunculus acris	5					Ranunculus ficaria	5
						Ranunculus repens	5
SHRUB							
Ulex europaeus	10	Crataegus monogyna	40	Crataegus monogyna	30	Crataegus monogyna	
Crataegus monogyna	25	Rosa spp.	10	Rosa spp.	10	Rosa spp.	30
Prunus spinosa	20	Sambucus nigra	5	Corylus avellana	5	Fagus sylvatica	15
Corylus avellana	5	Ulex europaeus	5	Fagus sylvatica	5	Fraxinus excelsior	5
Rosa spp.	5			Acer pseudoplatanus	5	Ilex aquifolium	5
				Prunus spinosa	5	Ulex europaeus	5
				Sambucus nigra	5		5

Table 2: Differences in species composition between the Vegetation Classes shown as % constancy. (Species with constancy <5% are not shown)

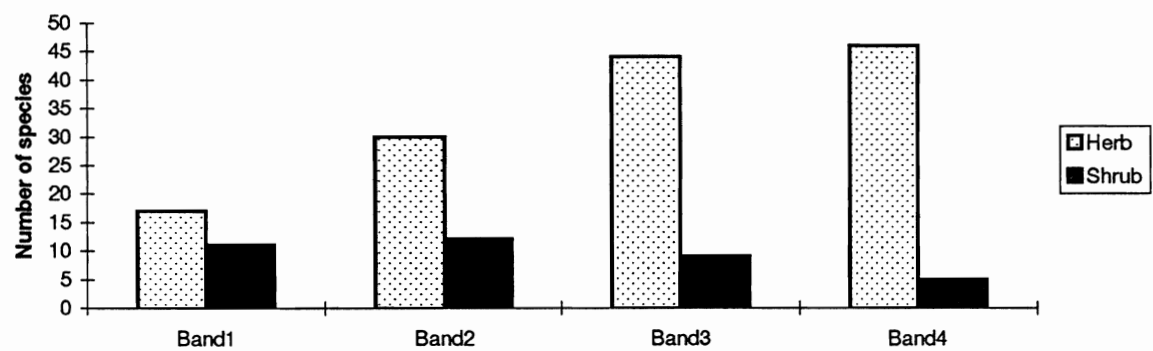
Figure 1: Number of herb and shrub species VC1



VC2



VC3



VC4

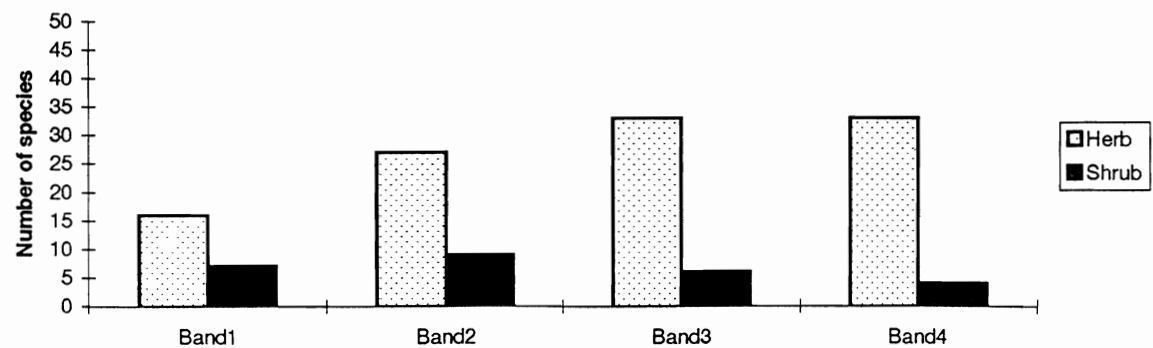


Figure 2: Ordination axis scores for bands showing mean values and standard errors

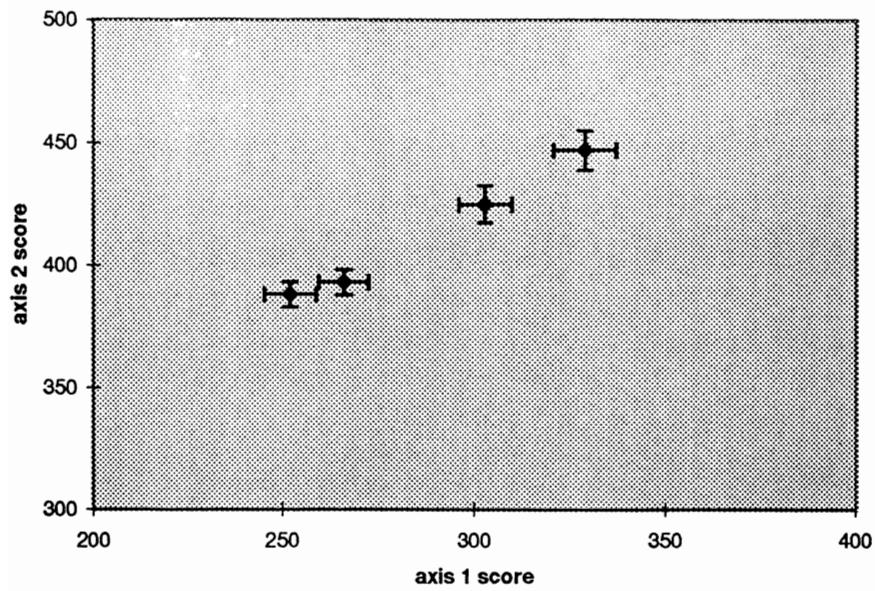


Figure 3: % composition of TWINSpan groups in terms of bands

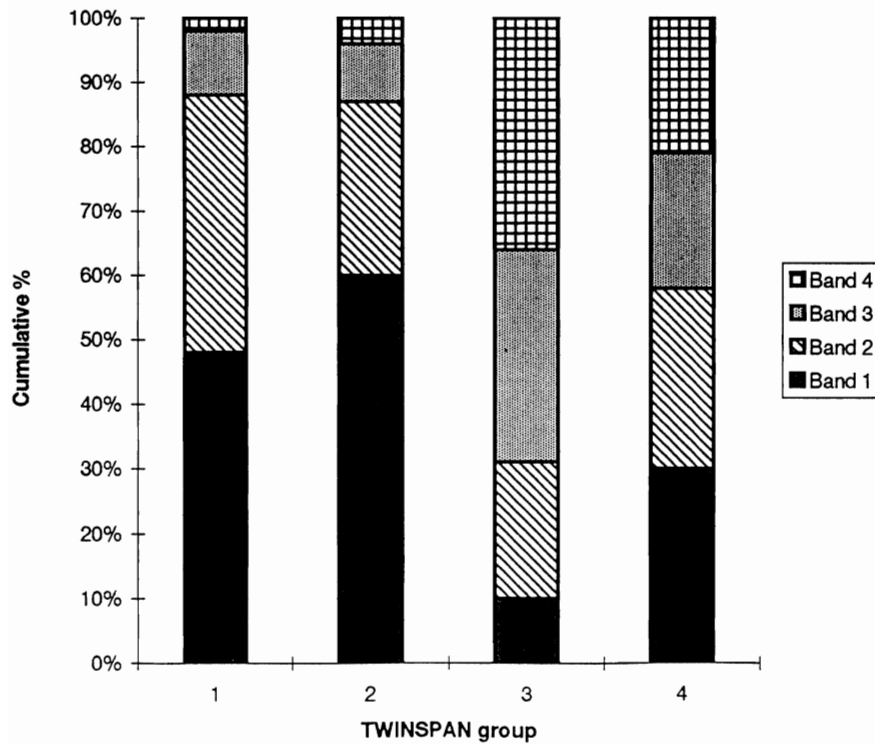


Figure 4 Mean Ellenberg indicator scores for light.

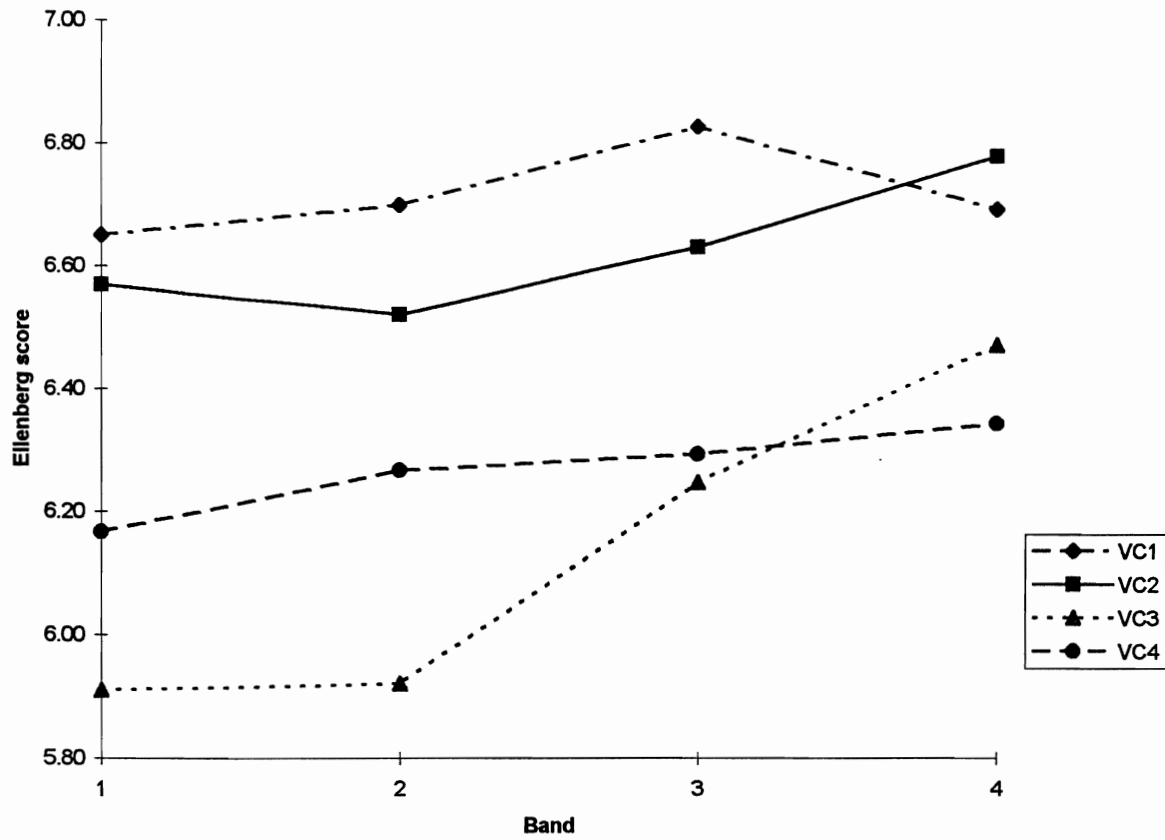


Figure 5. Mean Ellenberg indicator scores for moisture.

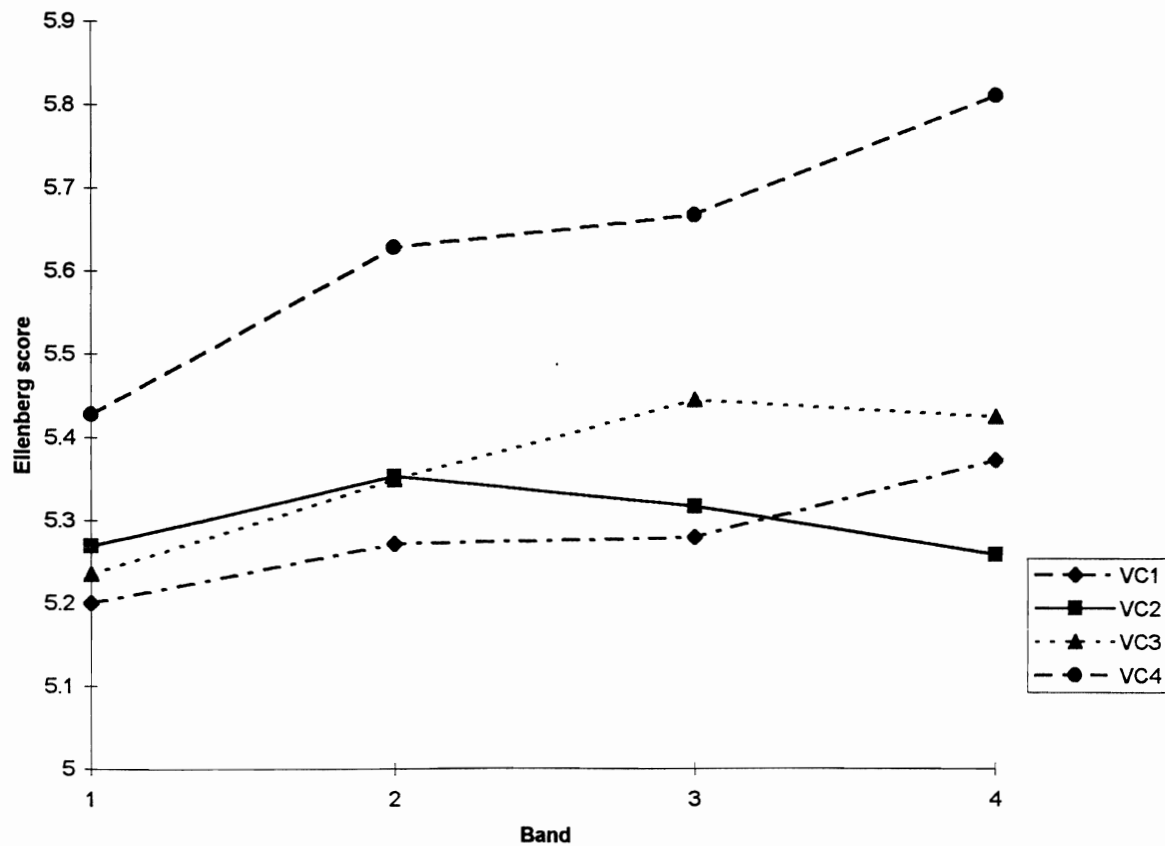


Figure 6 Mean Ellenberg scores for pH

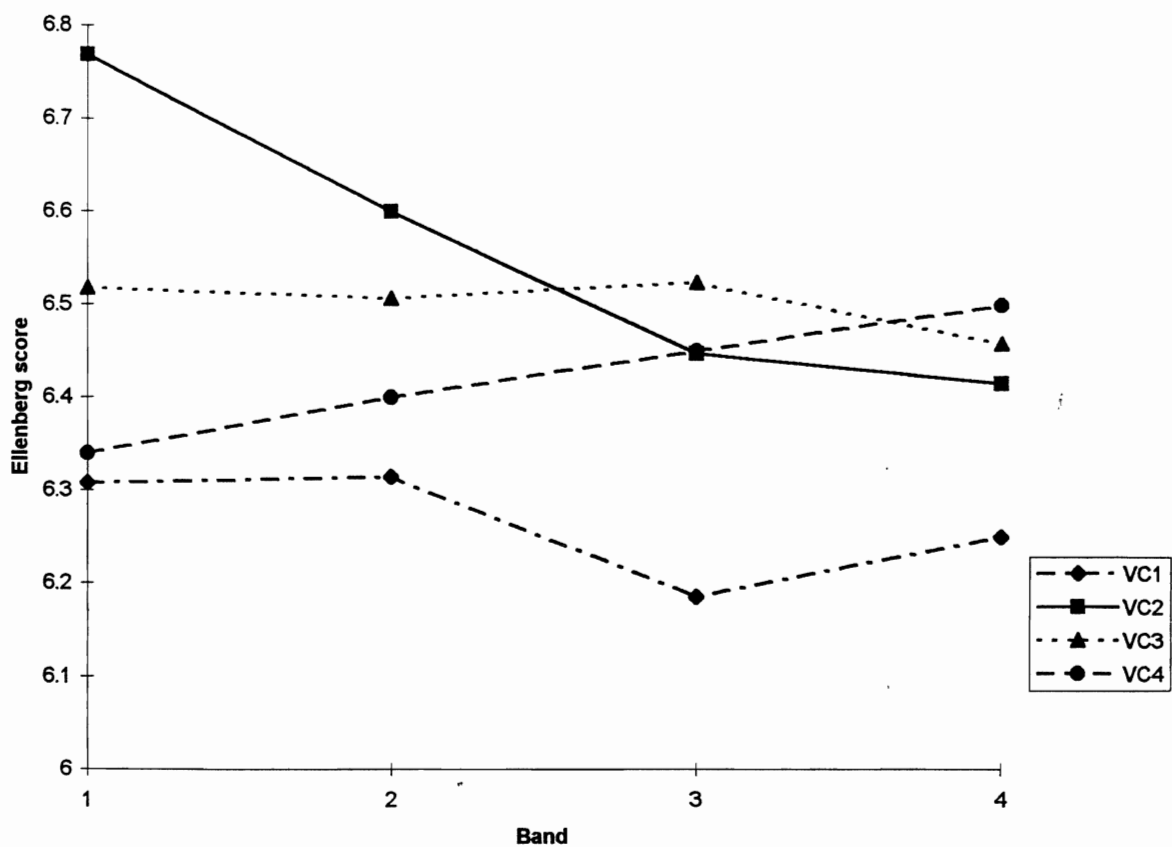
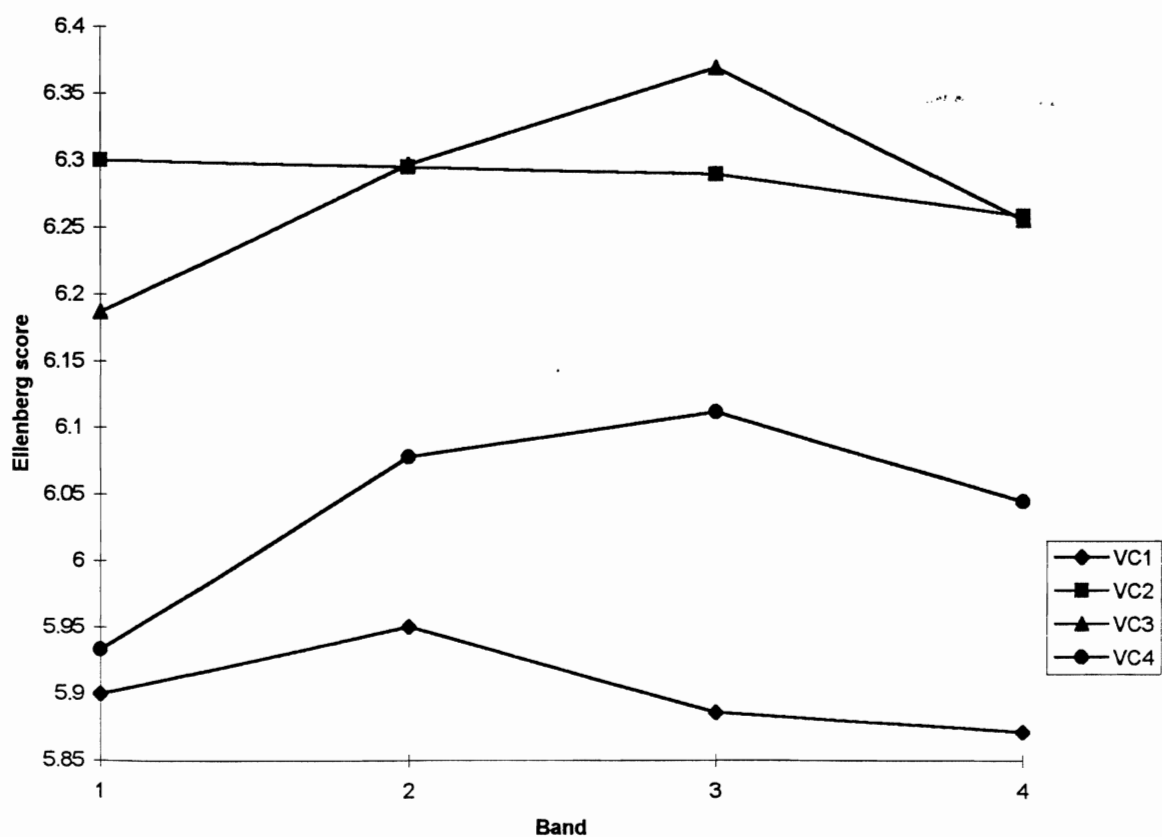


Figure 7 Mean Ellenberg indicator scores for fertility



Impact of changes in woodland canopy

1) Were increases in woody species associated with decline in species richness in lowland and upland woodland vegetation ? 2) Is there evidence that growth of conifers is linked to reduced species richness in upland woods ?

1) Methods & results

Because our interest was in the effects of increasing woodland cover between 1978 and 1990 we defined a subset of plots that were not classified as a woodland vegetation type in 1978 ie. neither aggregate class 5 or 6, but where all plots were in parcels classified by land cover survey code as woodland in 1978. In this way our starting point focusses upon vegetation set in a woodland context but where the plot vegetation itself was not classified on floristic grounds as a woodland assemblage. From this starting point we then analyse changes in species richness between 1978 and 1990 answering a series of questions as follows:

a) Did species richness change in plots that moved into a wooded vegetation type in 1990 ?

Year	Aggregate class	Land cover	Mean species count	<i>p</i>	<i>n</i>
1978	Not 5 or 6	Woodland	26.24	*	124
1990	Any	Woodland or not	23.52		

b) Did species richness change in plots that were not a wooded vegetation type in 1978 and were still not classified as woodland vegetation in 1990 ?

Year	Aggregate class	Land cover	Mean species count	<i>p</i>	<i>n</i>
1978	Not 5 or 6	Woodland	26.32	ns	85
1990	Not 5 or 6	Woodland or not	26.42		

c) Did species richness change in plots went from being a woodland vegetation type AND woodland land cover in 1978 into a non-woodland land cover in 1990 ?

Year	Aggregate class	Land cover	Mean species count	<i>p</i>	<i>n</i>
1978	Not 5 or 6	Woodland	29.28	ns	36
1990	Any	Not woodland	31.33		

d) Did species richness change in plots that were neither a woodland vegetation type OR land cover in 1978 but may have moved into both in 1990 ?

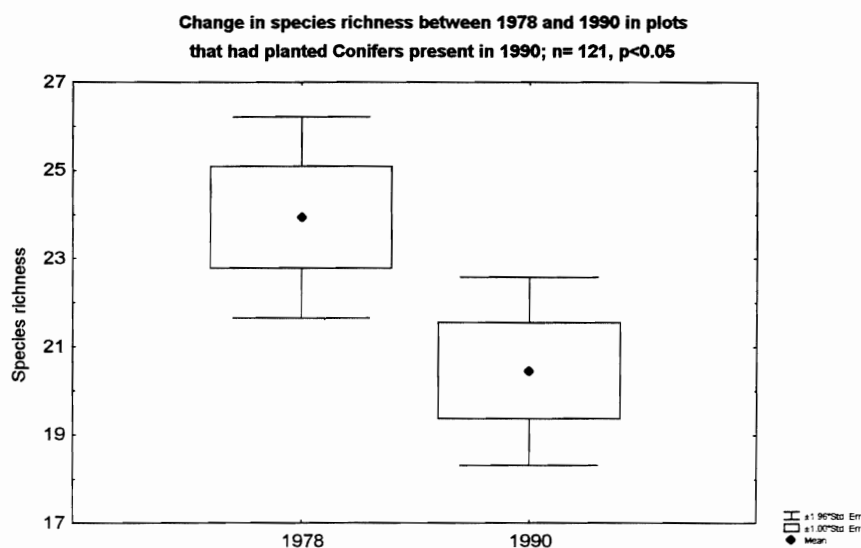
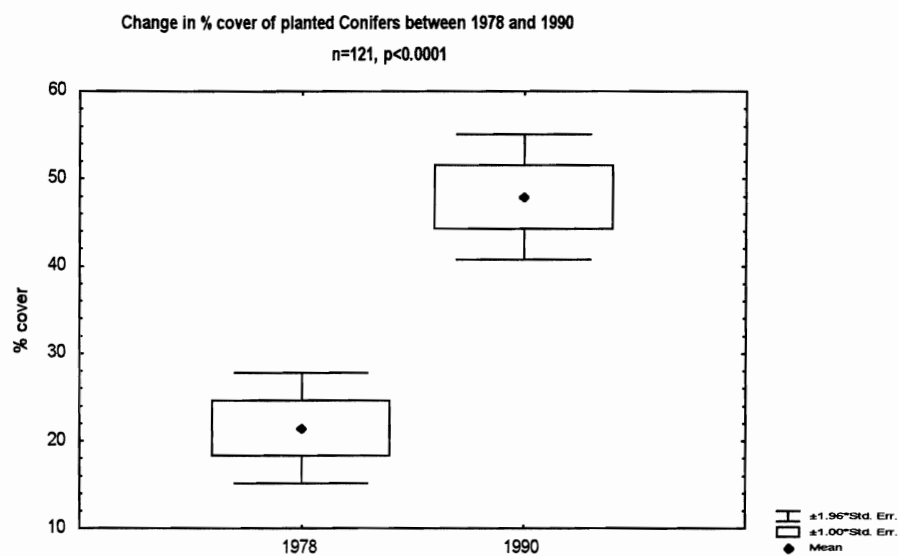
Year	Aggregate class	Land cover	Mean species count	<i>p</i>	<i>n</i>
1978	Not 5 or 6	Not woodland	24.85	ns	62
1990	Any	Woodland	24.27		

Conclusions

- The only significant result is a decline in species richness for those plots that moved from a non-woodland vegetation type in 1978 into a woodland type in 1990.
- The smaller sample sizes and so lessened power of the other tests should be borne in mind although these results suggest that where the influence of woody species at the plot and land cover level remains the same or decreases between years (b and c) then there is a lesser effect on species richness.

2) Methods & results

A subset of plots was selected such that there was an increase in all conifer species cover between 1978 and 1990. Within this data set analyses of change in cover and species richness (excluding conifers) were carried out using Wilcoxon matched pairs tests..



Changes in abundance of competitive species and their effects on species richness in Countryside Survey data

Part 1

Did widespread competitive species increase in cover between 1978 and 1990 across GB ? What happened to species richness in plots where these species were recorded in either year ? What happened to species richness in plots where these species increased between years ?

Method

Change in cover of seven species was analysed using only plots where cover in either 78 or 90 was at least 5% thus excluding data related to species dynamics and sampling error at low cover values. Species richness data included only taxonomically reliable category 1 species. Three analyses were carried out: Firstly changes in cover in all the listed species was analysed irrespective of the direction of change. Secondly, species richness changes were analysed over the same dataset. Thirdly changes in species richness were examined within only the smaller number of replicate plots in which each species increased in cover. This was done in order to focus specifically upon the effects of increasing species cover on species richness.

Significant differences in species richness and cover between years were analysed using a Wilcoxon matched pairs test.

Results

1) Changes in individual species cover

Species	Change	W_statistic		p
<i>Cirsium arvense</i>	+	2432.5	***	<0.001
<i>Elymus repens</i>	+	14946	***	<0.001
<i>Holcus lanatus</i>	+	87762	**	<0.01
<i>Lolium perenne</i>	+	121962	ns	
<i>Pteridium aquilinum</i>	+	4057.5	*	<0.05
<i>Rubus fruticosus</i>	+	10168.5	***	<0.001
<i>Urtica dioica</i>	+	16990	*	<0.05

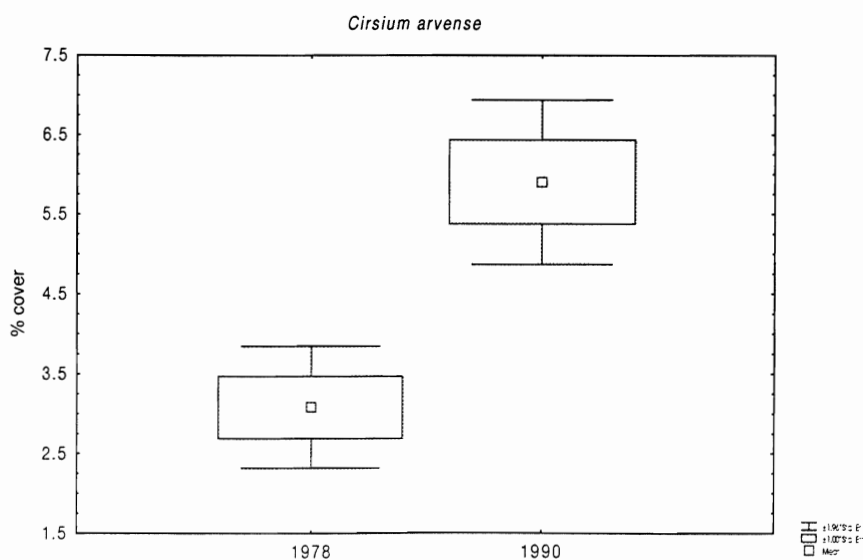
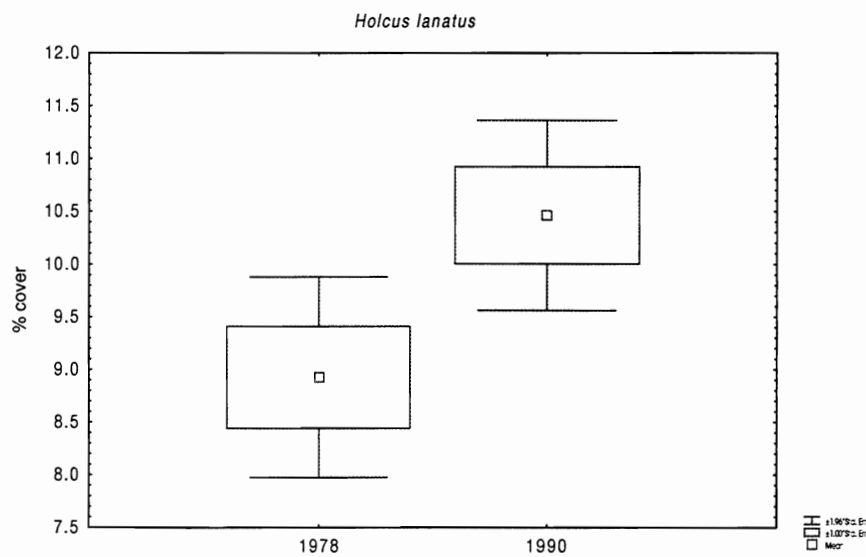
2) Changes in species richness in plots where each above species was recorded at at least 5% cover in either 1978 or 1990

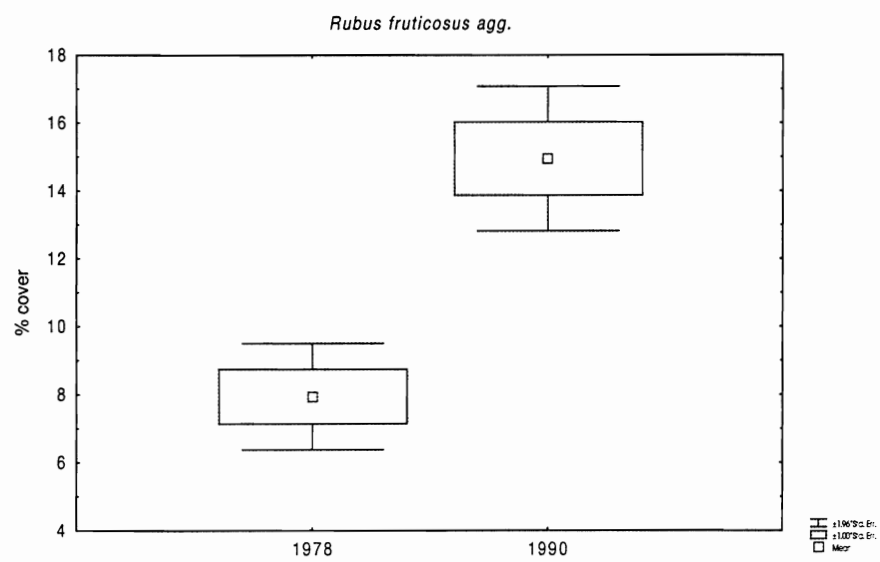
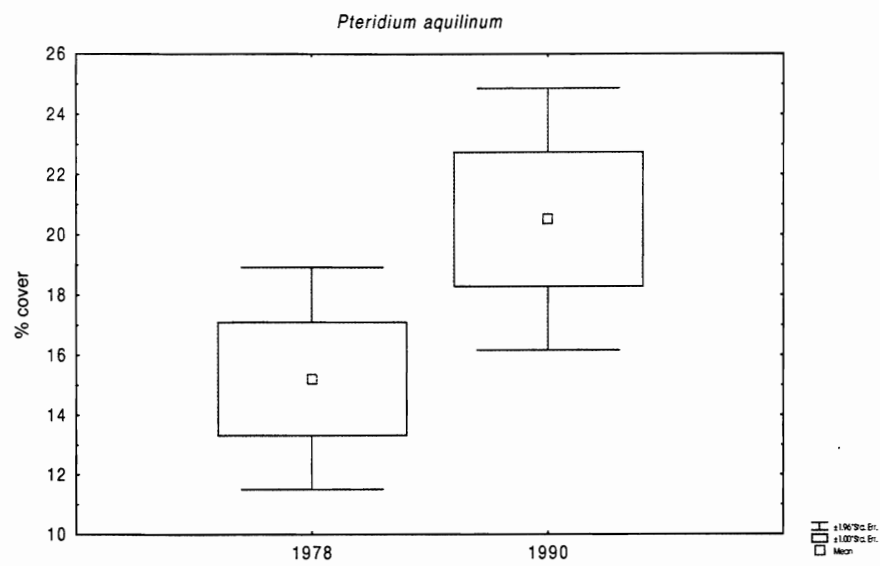
Species	Change	W_statistic	
<i>Cirsium arvense</i>	-	3485	ns
<i>Elymus repens</i>	+	20263	ns
<i>Holcus lanatus</i>	-	93862.5	ns
<i>Lolium perenne</i>	-	117381.5	ns
<i>Pteridium aquilinum</i>	-	4256.5	ns
<i>Rubus fruticosus</i>	-	15435	ns
<i>Urtica dioica</i>	-	19591	ns

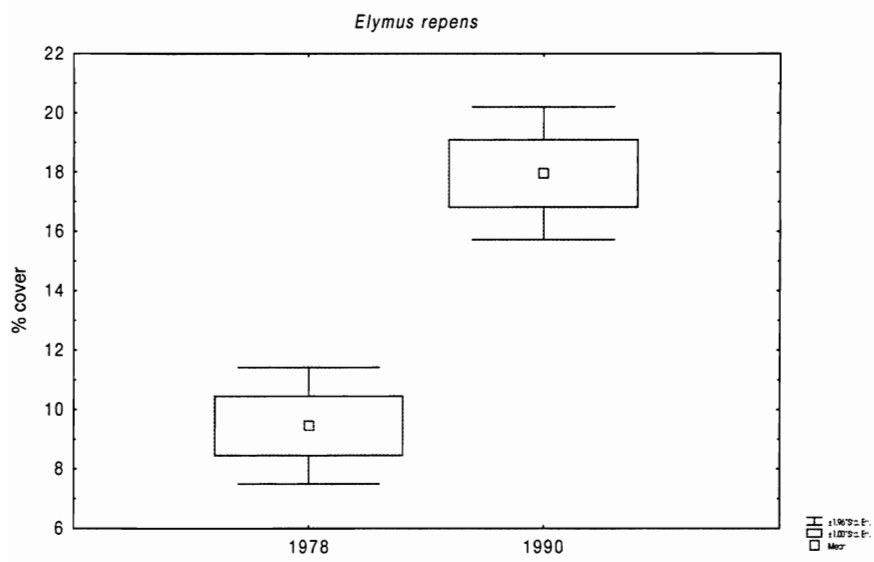
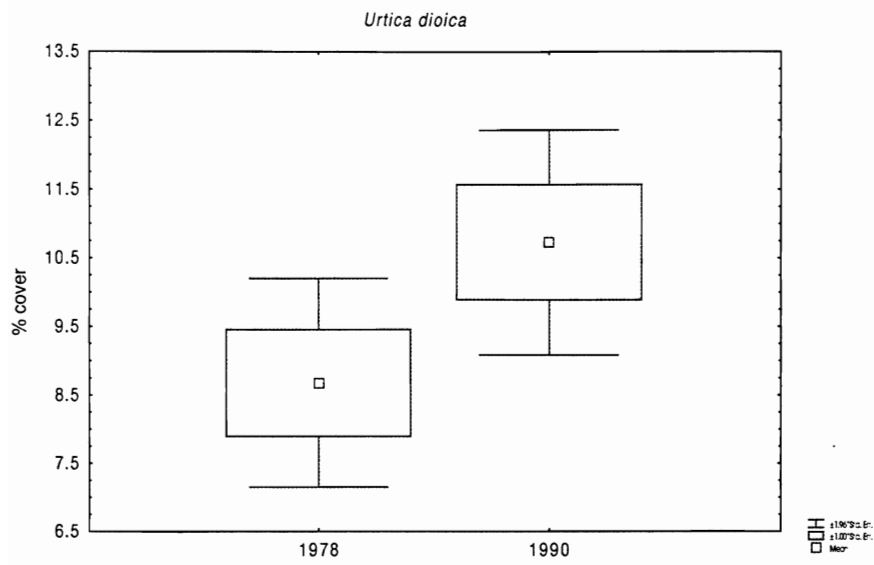
3) Changes in species richness in plots where each species was recorded at a minimum 5% cover in either 1978 and 1990 and increased in cover between 1978 or 1990.

Species	Change	W_statistic	
<i>Cirsium arvense</i>	+	1130.5	ns
<i>Elymus repens</i>	+	9175	ns
<i>Holcus lanatus</i>	+	28877	ns
<i>Lolium perenne</i>	-	32730.5	ns
<i>Pteridium aquilinum</i>	-	1573.5	ns
<i>Rubus fruticosus</i>	-	6314	ns
<i>Urtica dioica</i>	-	6459.5	ns

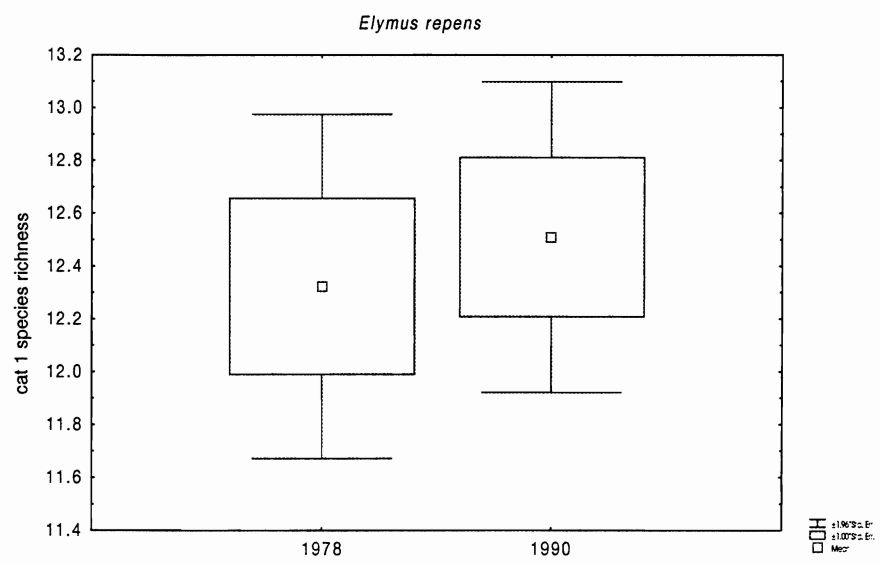
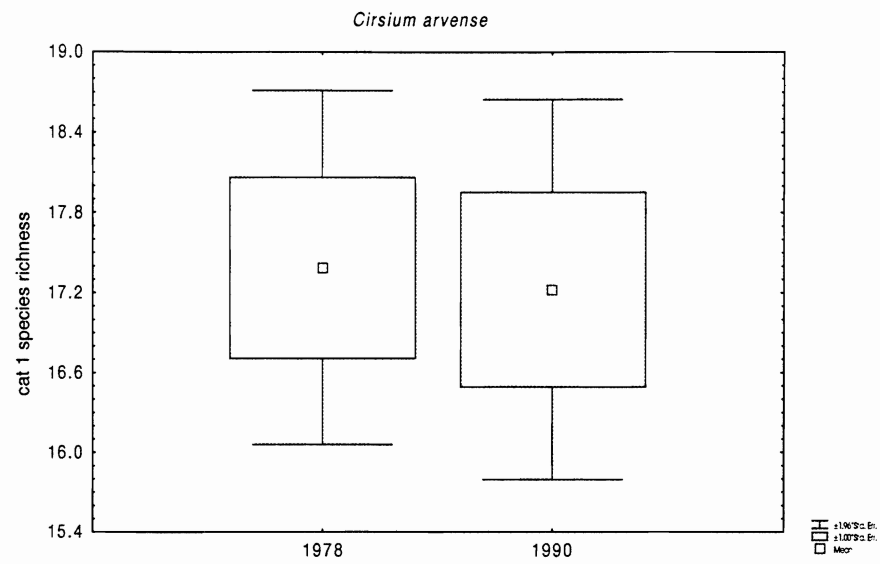
Change in cover

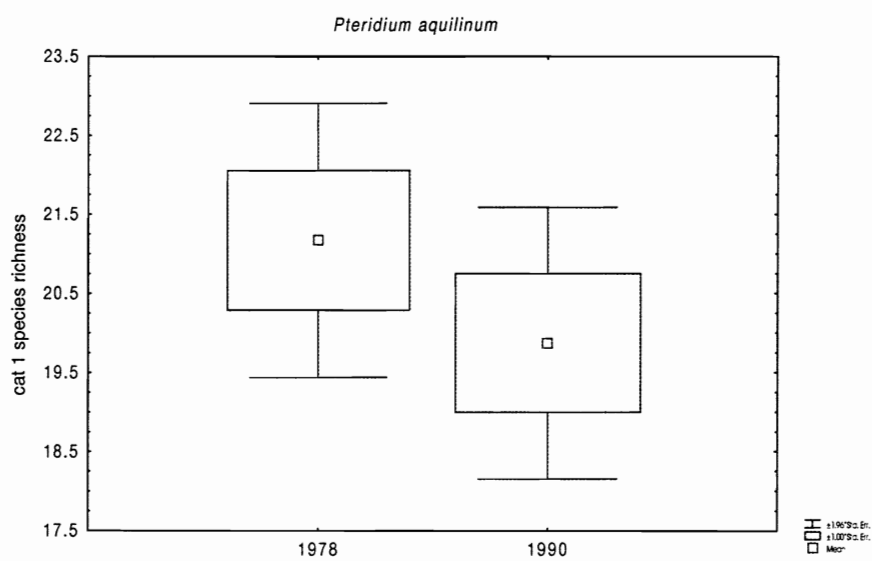
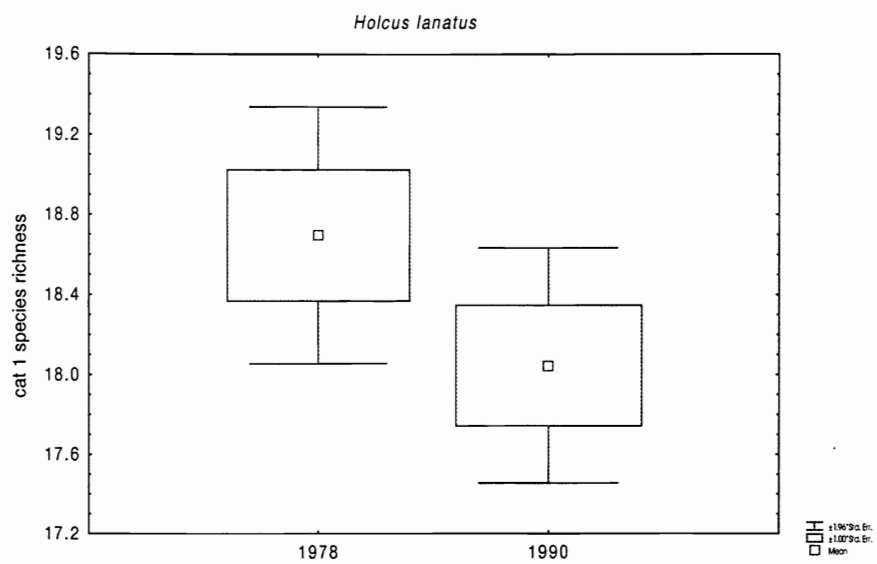


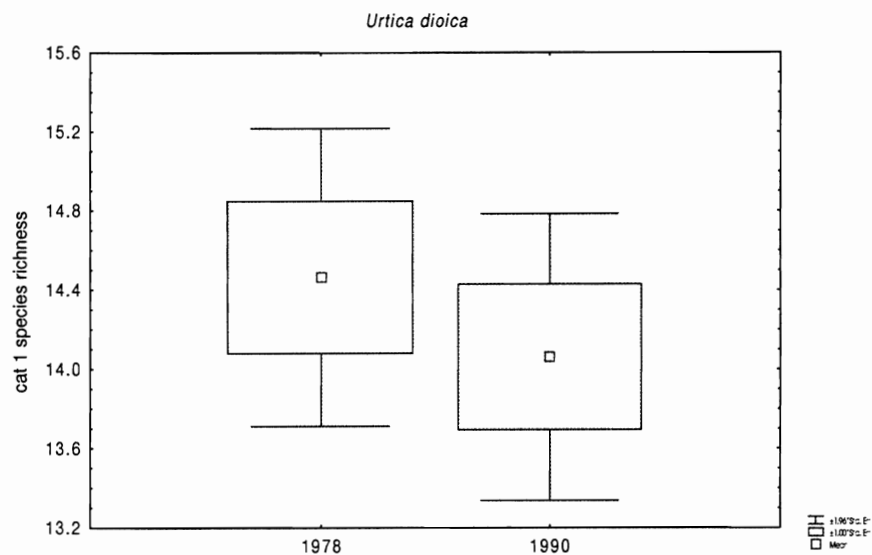
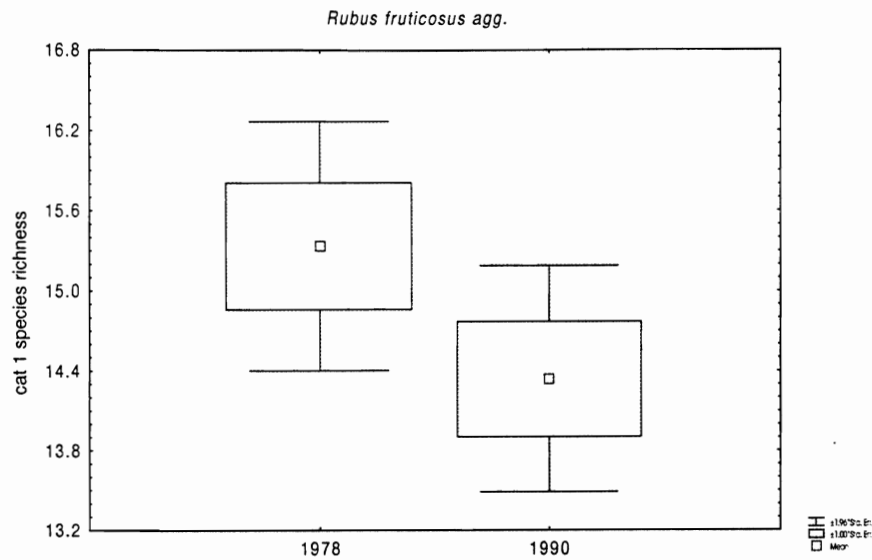




Change in species richness







Discussion

Despite statistically significant increases in cover of six out of seven species in all plots in which each species occurred, no significant reductions in species richness in these plots were detected at $p < 0.05$. However for all species other than *Cirsium arvense* and *Elymus repens*, the trend in species richness was downward whilst the p value for decline in species richness in plots with *Rubus fruticosus agg.* was 0.052. The conservative nature of the non-parametric test and the possibility of type II error should also be recognised. No significant changes in species richness were detected between plots in which the selected species increased only. Indeed in plots in which *Cirsium arvense*, *Elymus repens* and *Holcus lanatus* increased the trend was for increasing species richness. A possible explanation is that these tests were

using inappropriate variables. For example, on average more than one competitive species may act jointly within plots to depress species richness but only affecting populations within a functional subset of all those present ie. the least tolerant species, whilst other species less sensitive to the expansion of potential dominants may exhibit mixed responses even including expansion. To explore whether increases in competitive species richness had occurred, count data for only competitors was analysed by repeated measures GLM (full details are not reported here). Results indeed showed that mean increases in competitor richness occurred in many cells of the stratified data for example in road verges in the arable landscape and X plots in the marginal uplands.

Also on road verges and in hedgerows it is possible that all field layer species can succumb to shading by hedgerow shrubs and trees, irrespective of established strategy. However because of the mixed species composition of hedge lengths and the difficulties in estimating cover for these species it is less likely that significant associations between expansion of individual woody taxa and reduced species richness are detected. This is especially so if neglect and rising fertility are characterised by the establishment of other woody species which would contribute to increasing species numbers offsetting the decline in richness of the herbaceous component. In fact among woody species, only changes in cover of Conifers (grouped together) and *Acer pseudoplatanus* were significantly negatively associated with species richness changes (see below).

In the next section we search for the expected association between increase in cover of competitors and depressed species richness by summing cover changes and species richness changes over functional groups rather than individual species in the hope that increased sample size and the heightened sensitivity of the test will reveal expected patterns at the landscape scale.

Part 2

Do changes in cover of functional groups provide better correlates with changing species richness ? Do associations differ between landscape, plot type and aggregate class ?

Introduction

Changes in species richness detected in CS data in many cases involve decreases in less aggressive species characterised by a more stress-tolerant or ruderal established strategy (EcoFact report & draft Nature paper). In some situations this may be a response to an increase in competitors including trees, shrubs, herbs and grasses.

In terms of the causes of these changes, different management impacts at the patch and management unit scale may have the same net effect between patches. For example neglect of road verges, consequent litter build-up and increase in competitive plants may have the same effect on plant species richness as grassland improvement or conifer growth.

At large scales some effects may find a common denominator in terms of process ie. raised or lowered fertility and more or less disturbance but initial floristics, abiotic conditions and the availability of propagules can differ widely from site to site so that many different species can be implicated in competitive response and effect from patch to patch. Also in any particular patch the CS plot may incorporate too much heterogeneity leading to insensitivity to the dynamics of a subset of species within the plot. A consequence is that analysing the effects on species richness of individual competitive species may fail to reveal any effect (see part 1).

Since CS random samples take in much heterogenous vegetation (expressed within quadrats as well as between) the starting cover of many species in 78 will be highly variable. For example an increase of 20% cover of a species already present at high cover in 1978 may not be associated with a reduction in species richness if the critical episode of expansion and exclusion had already occurred.

The analyses that follow take into account plots where species declined in cover before 1990. However there is no way of knowing if the species was removed weeks or months before the 1990 survey. Therefore detection of any trend is necessarily based upon two snapshots separated by 12 years of inestimable dynamics. Redundancy and therefore robust analysis of associations is however, favoured by the large number of samples available covering the whole of GB.

Methods

Only records for taxa consistently and reliably identified to species level in the Countryside Survey were included in all analyses. Two sets of analyses were carried out on all replicate CS plots in all of which associations were sought between some measure of change in species richness between 1978 and 1990 and change in a measure of the total cover of a functional group of species. Species that had merely changed from being present at less than 5% cover to absent or *vice versa* between years were excluded from the analysis. Data were arranged as sets of 2-way tables stratified by landscape, plot type and aggregate vegetation class. The two types of association analysis were as follows:

1) Change in total cover of potentially dominant (PD) species versus change in richness of non-competitive (NC) species.

The response variable is the number of plots in which species classified as stress-tolerators or ruderals (NC species - figure 1.) increased or decreased. The division roughly corresponds to that in Grime (1985) and Stockey & Hunt (1994). We therefore partition within the response changes in that group of species likely to be most vulnerable to the individual competitive effect of species with a marked ability to capitalise on increased availability of resources as a result of a variety of management impacts (including dereliction) particular to the landscape element and the vegetation type. The design variable is the total change in cover of all potentially dominant species within each plot where the classification of potential dominants again corresponds largely to Grime (1985) and Stockey and Hunt (1994). In this way the sample size and sensitivity of each test is increased since a) the dynamics of individual species with a presumed common competitive effect are summed over all plots in which any one species occurred, and b) we focus on functional extremes where changes are theoretically most likely to show mutual dependence.

2) Change in total cover of potentially dominant graminoids[#] (PDG) versus change in richness of NC species.

The response variable is as 2). The design variable here includes only graminoid PD species importantly excluding all woody species. This is because in many plagioclimax communities the dominants are monocotyledons (often grasses) and there are numerous examples of deterioration in semi-natural communities characterised by increases in aggressive monocots and reductions in species richness (Smith & Rushton, 1994, Ward and Jennings, 1990, Dolman and Sutherland, 1992, ten Harkel and van der Meulen, 1996, Parr and Way, 1988, Willis, 1963, Willems et al., 1993, Mountford et al., 1993, Bobbink et al., 1988, Grootjans et al., 1996)

All associations between change in cover and change in species richness were tested by sequences of Fisher's exact tests. A negative phi coefficient indicates a negative relationship between changing cover and changing species richness. A positive sign indicates that cover and richness change in parallel.

Results

Total number of separate tests for 1) and 2) = 126. Therefore six significant results are expected by chance at $p=0.05$ and one significant result at $p=0.01$.

Analysis 1)

Detected associations are largely interpretable (table 1). When data was stratified by aggregate class only, a negative association was detected within the tall grasslands encompassing, rank, infrequently mown vegetation of road verges, hedge bottoms, field edges and stream banks many of which would be inaccessible to stock. The

[#] definition follows Welch & Scott (1995) comprising narrow leaved herbs including monocotyledonous forms such as grasses, sedges and rushes.

trend here is, as expected, for increasing potential dominant cover to accompany decreasing numbers of stress-tolerant or ruderal species. A positive association was detected in the upland wooded aggregate class almost certainly explained by the growth of young plantation forestry from 1978 to 1990. Since plantation conifers have no CSR data their changes are not incorporated within the behaviour of the design variable whilst their ability to successfully suppress all field layer components results in a parallel decline in both competitors and NC species (see part 3).

Most native trees and shrubs have CSR data and where their expansion is likely we would expect to find a negative association between PD cover and NC richness and this is what was found in hedgerow plots in the pastoral landscape where, as elsewhere in the GB lowlands, hedges were profoundly affected by a lack of management between 1978 to 1990 with increasing shade cast as trees and shrubs grew up (Barr et al., 1991). The same negative relationship applied across streamside plots in the arable landscape, an environment that over the last two decades has been increasingly affected by eutrophication, lack of bank management (ref) and most recently low flows in some rivers and streams (Mantle and Mantle, 1992).

The parallel trend between PD cover and NC richness in fertile grasslands of the pastoral landscape is more difficult to explain. Although 10% of plots that were fertile grassland in 1978 moved into cultivation in 1990, a trend that would be most likely to accord with the detected association, the majority of plots either remained as fertile grassland or moved into tall and infertile grassland classes.

Analysis 2)

Negative associations between changing cover of competitive graminoids and changing NC richness were detected within fertile and infertile grasslands. These are important results in that for both aggregate classes an increase in competitive graminoid cover is here clearly implicated in the process causing decline in numbers of the smaller, less aggressive and increasingly uncommon plants of species rich grassland. Increasing competitive graminoid cover is also implicated in reduced NC richness across all road verge plots, in all streamside plots in the arable landscape and in all field plots in the pastoral landscape.

The positive association detected in the upland wooded aggregate class is likely to be linked to the suppression of all field layer plant species as conifer canopies close.

Positive associations were also detected in all hedgerow plots, streamside plots in the pastoral landscape and all upland X plots. In hedgerows the detected trend is likely to be the corollary of the negative association detected in analysis 1): Since woody species are excluded from the design variable we suspect that the suppression of most species under the shade cast by unchecked hedge growth is responsible. The same process can be partly invoked to explain the parallel trend, in at least afforested, upland X plots.

Table 1. Sum of PD cover versus NC species richness.

n	Landscape	Aggregate class	Plot type	<i>p</i>	ϕ
208	-	2	-	**	-
109	-	6	-	*	+
75	Arable	-	Streamsides	**	-
91	Pastural	-	Hedgerows	*	-
94	Pastural	2	-	**	-
107	Pastural	3	-	**	+

Table 2. Sum of PDG cover versus NC species richness.

n	Landscape	Aggregate class	Plot type	<i>p</i>	ϕ
426	-	3	-	*	-
501	-	4	-	*	-
154	-	6	-	*	+
111	Arable	-	Streamsides	*	-
136	Pastural	-	Streamsides	*	+
382	Pastural	-	Fields/unenclosed	**	-
292	Upland	-	Fields/unenclosed	*	+
256	-	-	Hedgerows	*	+
430	-	-	Road verges	**	-
111	Pastural	3	-	**	-

Conclusions

- Meaningful associations between functional groups of potentially dominant and non-competitive species can be detected in the wider landscape using CS data.
- The use of two response variables one including woody species and one comprising only graminoids appear to serve as complementary and interpretable approaches to the landscape scale effects of the processes favouring increases or decreases in cover of competitors.

Figure 1. The distribution of three divisions in the CSR triangle after Grime (1985). PD = Potentially dominant species comprise competitive dominants, ruderal dominants and stress-tolerant dominants. NC = Non-competitive species unable to coexist with PD species and adapted to extremely disturbed and/or unproductive conditions.

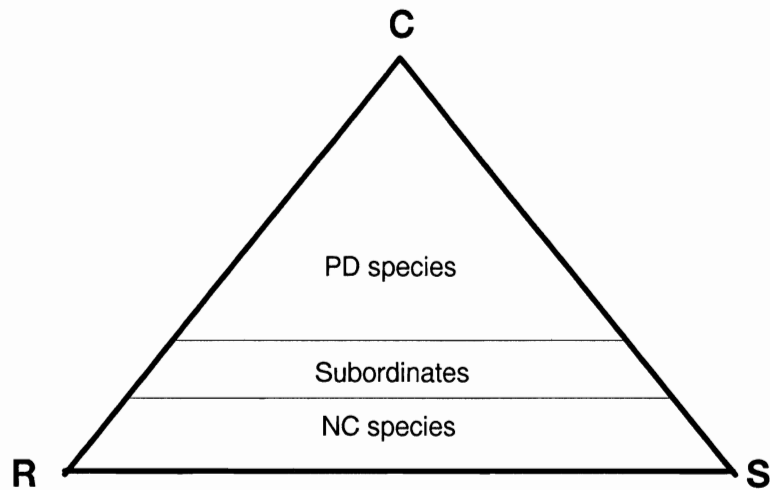


Table 2. Strategic composition of the groups used to define the design and response variables for association analysis. After Grime (1985).

Established strategy	Design variable (change in total cover per plot)	Response variable (change in number of species per plot)
R		✓
R/SR		✓
SR		✓
S/SR		✓
S		✓
R/CR		
R/CSR		✓
SR/CSR		✓
S/SC	✓	
S/CSR		✓
CR	✓	
CR/CSR		
SC	✓	
SC/CSR		
CSR		✓
C/CR	✓	
C/SC	✓	
C/CSR	✓	
C	✓	

Part 3) Species cover and Ellenberg scores

Associations between changing species cover and increase or decrease in Ellenberg fertility score were investigated by analysis of 2-way tables where a Fisher's exact test was carried out for each species with sufficient count data. Out of a total of 79 valid tests 10 significant results were detected at $p < 0.05$, 2.5 times as many as expected by chance (table 1).

A negative phi coefficient indicates a negative association ie. increasing species cover paralleled by decreasing Ellenberg score or *vice versa*. Nearly all associations are expected for example increases in *Galium aparine*, *Rumex obtusifolius* and *Urtica dioica* linked to increases in Ellenberg fertility score whilst decreases in Ellenberg score are linked to increases in species of unimproved grassland such as *Cynosurus cristatus*, *Agrostis capillaris*, *Trifolium pratense* and *Plantago lanceolata* or upland vegetation such as *Eriophorum vaginatum*, *Calluna vulgaris* and *Trichophorum cespitosum*. Since cover changes which involved a shift from absent to at least 5% or *vice versa* were excluded from the design variable, the results shown indicate that the richness of low or high fertility species is probably linked, at least in some circumstances, mechanistically to changes in cover of potentially dominant species in the sward. It is impossible to be more certain of the causal relationships involved although they are likely to include direct competitive exclusion of low fertility species with dereliction and/or eutrophication. In situations where increases in fertility score and species cover occur together it is likely that these are two aspects of a common response to favourable conditions for example local colonisation of low fertility substrates either spontaneously or by seeding eg. some road verges and set-aside on marginal agricultural land.

Table 1. Significant associations between changing species cover and changing unweighted Ellenberg fertility score.

Species	n	ϕ	Sig
<i>Agrostis capillaris</i>	579	-0.146	**
<i>Anthriscus sylvestris</i>	74	0.298	*
<i>Bellis perennis</i>	54	-0.449	*
<i>Calluna vulgaris</i>	378	-0.269	***
<i>Cynosurus cristatus</i>	193	-0.163	*
<i>Elymus repens</i>	322	0.142	*
<i>Eriophorum vaginatum</i>	130	-0.204	*
<i>Galium aparine</i>	137	0.301	**
<i>Heracleum sphondylium</i>	99	0.239	*
<i>Lolium perenne</i>	748	-0.079	*
<i>Molinia caerulea</i>	292	-0.208	**
<i>Plantago lanceolata</i>	135	-0.352	***
<i>Rumex obtusifolius</i>	68	0.389	***
<i>Sambucus nigra</i>	88	0.251	*
<i>Trichophorum cespitosum</i>	72	-0.355	***
<i>Trifolium pratense</i>	35	-0.417	*
<i>T. repens</i>	463	-0.175	**
<i>Urtica dioica</i>	310	0.306	***
<i>Vaccinium myrtillus</i>	91	-0.251	*

Examination of extreme plot Ellenberg scores

Two weeks were spent in the field in August 1997 to examine plots that had been identified as having either extremely high or low Ellenberg scores. The approach was discontinued after this period because it had been established that extreme individual plots always involved unique events and therefore could not be used by definition to assess changes represented by mean values.

Comparison of tissue nutrient levels with Ellenberg fertility values

The tissue levels of potassium, phosphorus and nitrogen of the herbaceous parts of a range of species were correlated with their Ellenberg value for fertility, hence referred to as N value. Table 1 shows that there are significant correlations between the Ellenberg N value of a species and tissue levels of potassium and phosphorus but not with nitrogen.

Table 1: Correlation coefficients (Spearman's R) of tissue nutrient concentrations with Ellenberg N values (n=29)

% Nutrient (dry weight)	R	p-level
% Potassium	0.486	<0.01
% Phosphorus	0.544	<0.01
% Nitrogen	0.319	n.s.

A principal components analysis (PCA) was carried out on the nutrient levels for the individual species in order to summarise the data and identify the gradients present. Potassium and phosphorus have the highest weighting on PCA1 and nitrogen has the highest weighting on PCA2 (Table 2) which suggests that there are two different aspects of fertility that may be operating.

Table 2: Loadings of nutrient variables on the first two principal components

	PCA1	PCA2
% Potassium	-0.745	-0.526
% Phosphorus	-0.881	0.054
% Nitrogen	-0.392	0.878
Eigenvalue	1.485	1.051
Proportion of total variance explained	0.495	0.350

The component scores for the individual species (Table 3) were then correlated with the Ellenberg N value. The first component, which is related to potassium and phosphorus status is significantly correlated with Ellenberg N value (Table 4), with a higher value of R and level of significance than the correlations with the individual nutrients.

Table 4: Correlation coefficients (Spearman's R) of component scores with Ellenberg N value (n=29)

	R	p-level
PCA 1	-0.665	<0.001
PCA 2	-0.020	n.s.

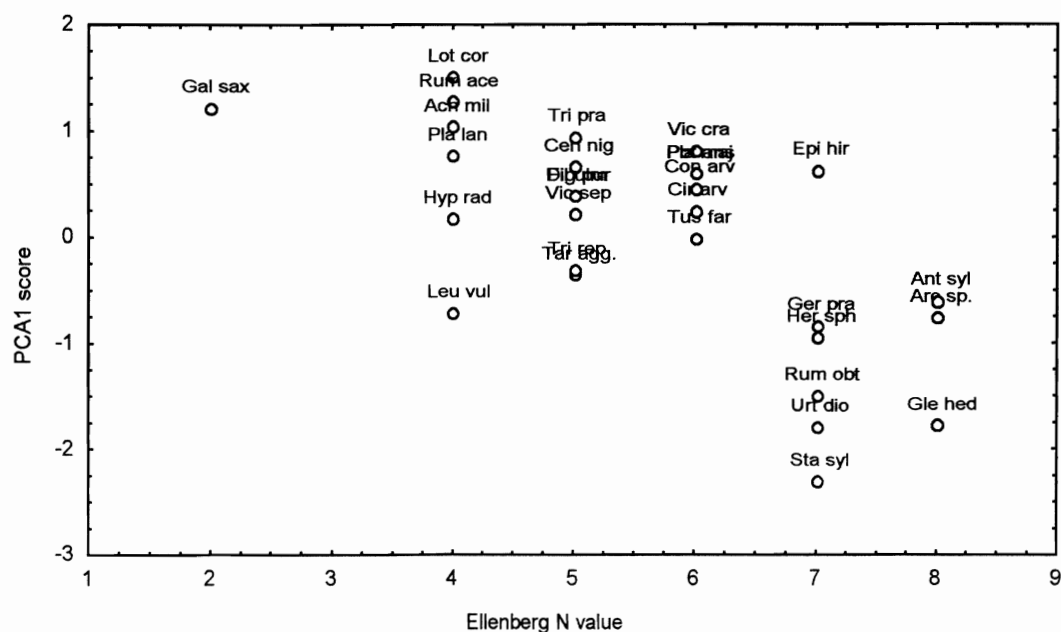
The relationship between PCA1 score and Ellenberg N value for the species is plotted in Figure 1, which shows that species which have higher Ellenberg N values have lower scores on the first component and *vice versa*. These species such as, *Rumex obtusifolius*, tend to be associated with more fertile conditions in contrast to species such as *Lotus corniculatus* which have higher scores on the first component and lower Ellenberg N values and are associated with less fertile conditions.

The PCA component scores can thus be viewed as measures of overall fertility in summarising the values for the individual nutrients. The above results suggest that for the species considered here, the Ellenberg N value can be better considered as reflecting the overall fertility status of a species, particularly with regards to potassium and phosphorus, rather than that of individual nutrients.

Table 3: Scores of the species on the first two principal components

	PCA1	PCA2
<i>Achillea millefolium</i>	1.037741	-0.37613
<i>Anthriscus sylvestris</i>	-0.59807	-1.60936
<i>Arctium minus</i>	-0.75896	-0.65772
<i>Centaurea nigra</i>	0.663259	-0.76628
<i>Cirsium arvense</i>	0.248508	0.233119
<i>Convolvulus arvensis</i>	0.461102	-0.01623
<i>Digitalis purpurea</i>	0.383731	-0.47119
<i>Epilobium hirsutum</i>	0.632524	1.532525
<i>Filipendula ulmaria</i>	0.386944	-0.10291
<i>Galium saxatile</i>	1.22317	-0.16235
<i>Geranium pratense</i>	-0.83317	0.858247
<i>Glechoma hederacea</i>	-1.78035	0.189275
<i>Heracleum sphondylium</i>	-0.94959	-1.02114
<i>Hypochaeris radicata</i>	0.172633	-0.43766
<i>Leucanthemum vulgare</i>	-0.71006	-0.72943
<i>Lotus corniculatus</i>	1.507034	0.302197
<i>Plantago lanceolata</i>	0.762146	-0.71979
<i>Plantago major</i>	0.592171	-0.8817
<i>Potentilla anserina</i>	0.592804	-0.48751
<i>Rumex acetosella</i>	1.272071	-0.23406
<i>Rumex obtusifolius</i>	-1.49956	0.682568
<i>Stachys sylvatica</i>	-2.30745	-0.93242
<i>Taraxacum officinale</i>	-0.35836	-0.71918
<i>Trifolium pratense</i>	0.944907	0.921105
<i>Trifolium repens</i>	-0.31082	1.626028
<i>Tussilago farfara</i>	-0.01357	-1.44266
<i>Urtica dioica</i>	-1.79111	2.093964
<i>Vicia cracca</i>	0.815334	1.570773
<i>Vicia sepium</i>	0.214972	1.757918

Figure 1: The relationship between PCA1 score and Ellenberg N value ($R = -0.665$)



The impact of pollution on biodiversity

Introduction

The question of the impact of pollution on biodiversity is extremely broad. There are many recognised pollutants, often chemicals and compounds which occur naturally and may even be essential for life, but which can be damaging at concentrations beyond normal. For this module, one form of pollution, nitrogen based compounds, was identified as an example and its relationship with the Great British flora examined. The work was done in part as a desk study, but also samples were collected in the field to examine the variation in levels and identify any evidence for changes in floristic diversity that have been observed.

Nitrogen is essential for all life and in many habitats it is the nutrient which limits plant growth. Different compounds containing nitrogen form part of the natural atmosphere and vary in concentration in both time and space; man's activities increase and distort both spatial and temporal distributions. Nitrogen is scarce in the mineral components of soil, and is mainly derived from organic material and atmospheric deposition. Some organisms can fix nitrogen from air (eg some lichens and mosses) and soil bacteria associated with leguminous plants can trap over $100 \text{ kg N ha}^{-1} \text{ yr}^{-1}$. Traditional agricultural practice used nitrogen fixers such as clover and alfalfa to restore the soil nitrogen balance and replace that removed during harvest. The changes in agricultural management over recent decades have led to nitrogen based fertiliser being applied; which may well be more than double that which could have naturally been fixed.

Changes in botanical biodiversity due to atmospheric nitrogen deposition can occur through a number of possible mechanisms. The fertiliser effect may alter the competitive balance of vegetation encouraging aggressive rapid growing species; nitrates may occur at toxic levels; the physiological changes which allow tolerance may manifest themselves as increased sensitivity to other components of the system; the nitrification of atmospheric ammonium in the soil is an acidifying process. Within this project it will not be possible to identify the mechanisms of change.

Sampling for the module was carried out in 1996 and 1997 and was split between the uplands, where ericaceous shrubs and mosses were collected, and the lowlands where hawthorn from hedgerows was sampled.

Upland studies

Racomitrium lanuginosum has been described as being sensitive to increases in atmospheric nitrogen. Observation and experimental evidence suggest that *R. lanuginosum* declines where nitrogen levels have risen, probably through increased growth of competing grasses and sedges. The initial objective was to investigate levels of nitrogen in *R. lanuginosum* and in plants in similar habitats in areas where it would be expected to be found. *R. lanuginosum* is perennial and its growth form adds leaflets to existing fronds; to reduce the variation in nitrogen levels in plants, only the current years growth was taken and analysed. Where *R. lanuginosum* was absent, the years growth from *Sphagnum* and/or *Calluna vulgaris* was collected.

Sites for sampling were identified from quadrat data of the Countryside Surveys in 1978 and 1990. *R. lanuginosum* was recorded in 457 quadrats in 103 sample squares ranging across 17 land classes during Countryside Survey 1990 (Table 1). The sample squares were predominantly in the marginal uplands and uplands with a fair sprinkling on the outer isles. In order to keep within budget, the island squares were not sampled. Where possible sites were selected that contained more than one quadrat in which *R. lanuginosum* was recorded. Sites where *R. lanuginosum* was recorded in 1978 but not in 1990 were also selected for samples of *Sphagnum* and/or *C. vulgaris*. During the survey, if there were problems obtaining permission to sample, or *R. lanuginosum* could not be found, immediately adjacent sample sites were used.

Table 1 The frequency of sample sites in different land classes (Land classes in which *R. lanuginosum* was recorded in 1990, but which were not resurveyed are 5, 13, 15, 25, 27, 28, 29, 30, 31 and 32)

Land class	No. of sites in CS1990	No of quadrats in CS1990	No. sampled in 1996
17	4	7	3
18	6	23	2
19	1	2	2
21	16	92	9
22	5	9	4
23	9	47	4
24	15	103	8

Samples of *R. lanuginosum*, *C. vulgaris* and *Sphagnum* were taken in autumn 1996 so as to include as much of the seasons growth and accumulation as possible. They have been chemically analysed for percent nitrogen. *C. vulgaris* and *Sphagnum* were also sampled when found in the same location as *R. lanuginosum* as they are considered slightly more tolerant of increased nitrogen. If higher levels of nitrogen were found in the samples where *R. lanuginosum* was absent, it would be suggestive that atmospheric pollution could be linked to the change.

When the samples were taken the sites were also assessed for grazing pressure, as there may be an interaction between grazing and pollution on biodiversity. Grazing animals play a complex role in modifying the quantity of nitrogen being removed from and passed into the plants and soil; grazing has also been shown to have a controlling influence on biodiversity. The grazing and interaction between grazing and atmospheric deposition is reported in the uplands module.

Samples were collected from 67 points in 26 sites in GB (Figure 1) with 42 of the sample positions contained *R. lanuginosum*, 62 contained *Sphagnum* and 34 contained *C. vulgaris*. Figures 2, 3 and 4 show the arithmetic mean of the nitrogen content of samples within the individual species. In all three cases lower concentrations can be seen in the samples taken in the north which agrees with the variation reported by Baddeley *et al.*, 1994.

For all three species, there appears to be a strong correlation between percent nitrogen in the years growth and the total nitrogen deposition for the years 1992-94 (Table 2 and Figure 5). The deposition dataset being used for the comparison is the most recently produced and has been expressed at a 5 km square resolution, which is more detailed than the usual 20 km datasets. Total nitrogen has been interpolated from the sum of both NO_x and NH_y samples. Significant correlations can also be seen for location (lower in the north) which reflect a variety of confounding effects, including mans activities.

Table 2 Correlation between total N deposition in the years 1992-94 interpolated at a 5 km square resolution and the percentage N of the 1996 growth of different species.

Species	Correlation coefficient (Pearsons)	sample size	significance
<i>R. lanuginosum</i>	0.690	42	***
<i>C. vulgaris</i>	0.553	34	***
<i>Sphagnum</i>	0.733	62	***

Figure 1 Survey sites (1996) on total N deposition (1992–1994) surface

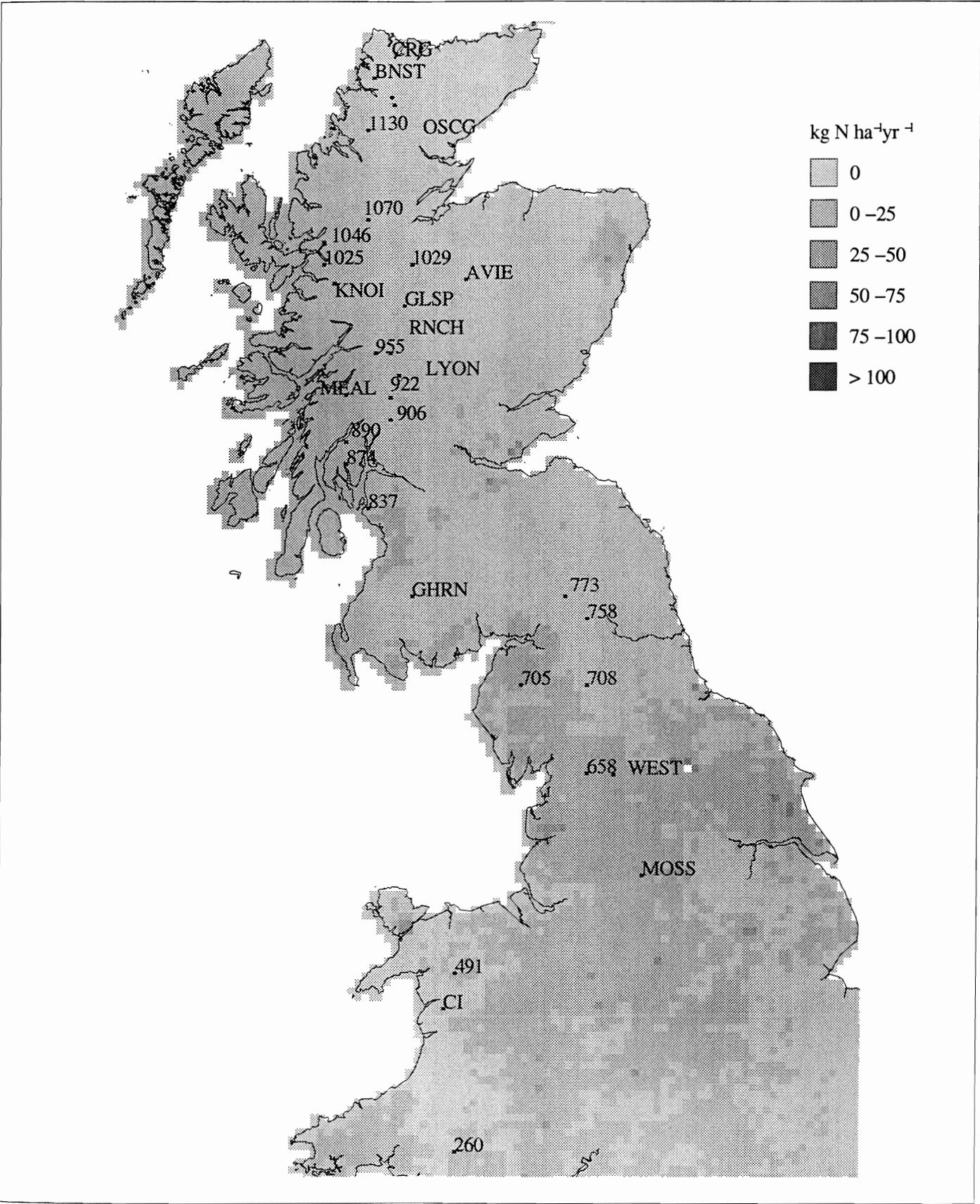


Figure 2 Variation in mean percent nitrogen of *Racomitrium lanuginosum* sampled in autumn 1996

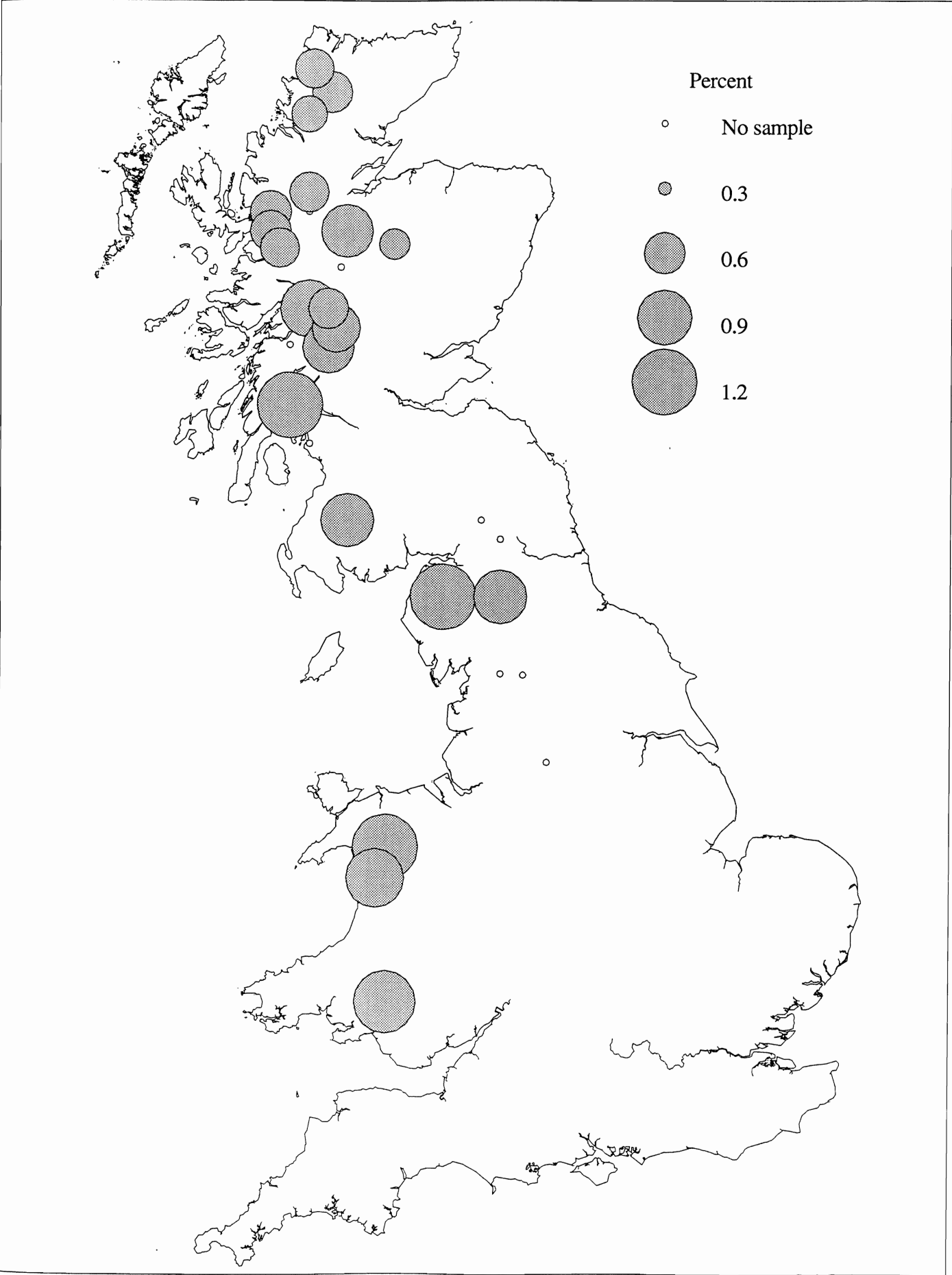


Figure 3 Variation in mean percent nitrogen of *Sphagnum* sampled in autumn 1996

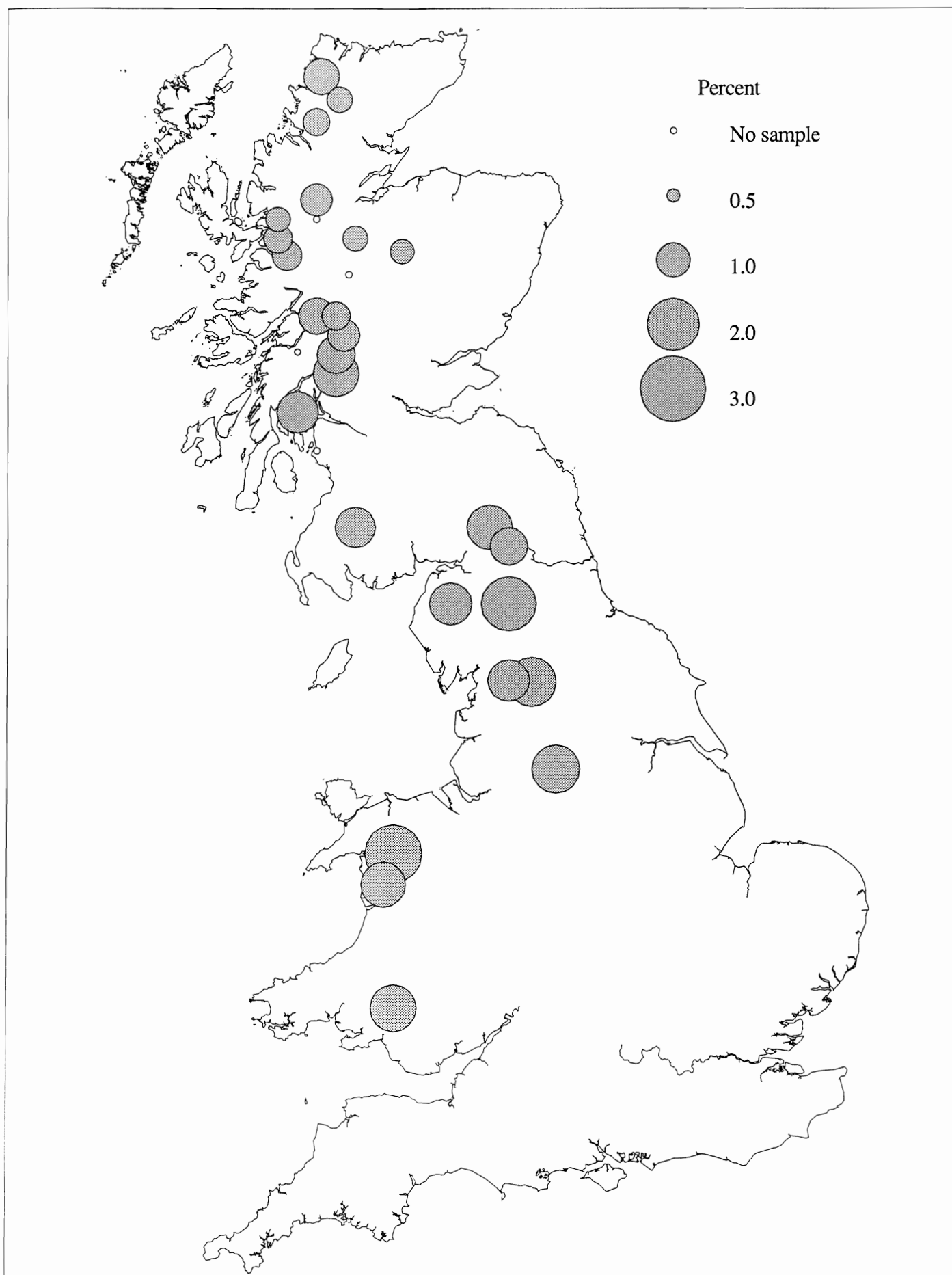
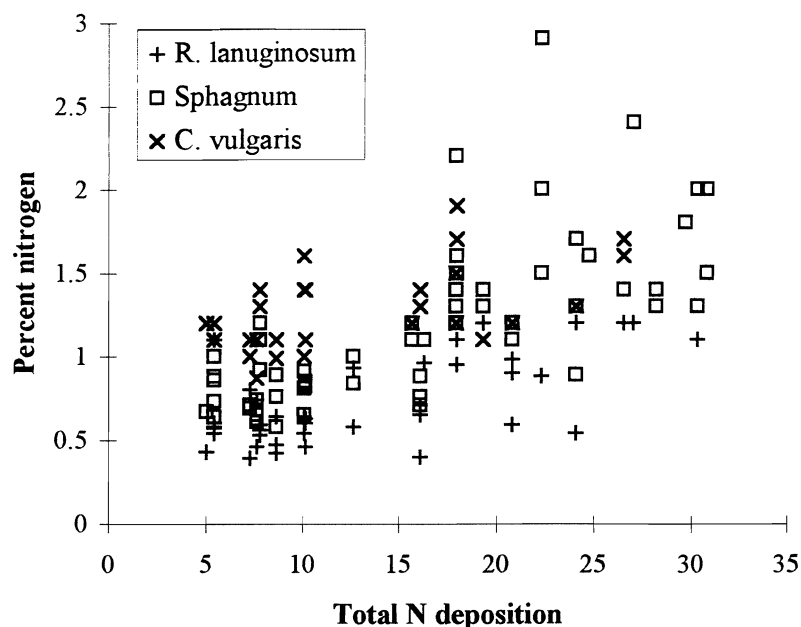


Figure 5 The relationship between total N deposition 1992-94 and nitrogen content of growth of species sampled in 1996.



The average levels found were at a similar level to those published by (Baddeley *et al.*, 1994), although they were describing *Racomitrium* heath, which is a rare habitat type and not the only place where *R. lanuginosum* is found. However, the levels differ from those previously recorded by the ITE chemical analysis section (11.0 mg g⁻¹) with the values being only two thirds of those reported in (Allen 1989); the levels reported for both *C. vulgaris* and *Sphagnum* agreed with the 1996 samples. In his thesis (Baddeley 1991) reported changes in the levels on the summit of Ingleborough, North Yorkshire spread over a century which show a nearly threefold increase in levels (from 4.6 mg g⁻¹ in 1879 to 12.3 mg g⁻¹ in 1989). The variation was much greater than the sample variation for each time point over the century (which had standard errors ranging from 0.2 mg g⁻¹ to 0.5 mg g⁻¹). The variation in *R. lanuginosum* recorded in 1996 shows a normal distribution about the mean (7.4 mg g⁻¹) with a standard error of (0.4 mg g⁻¹).

Micro-habitat may influence persistence and occurrence of species more than macro climate and other conditions. Although national levels may be correlated with change, the triggers which stimulate change may work at extremely local levels. Over a small area climate and soils can change dramatically for example the difference between the top of a hummock and the neighbouring dips.

The results from the national survey cannot prove the causes of any change. However, they do present estimates of change in vegetation and associated landscape features which can be used to suggest relationships. Unfortunately, changes that have occurred are now literally history and causes cannot be proved, the data should prove valuable in helping to formulate and sharpen hypotheses. The only real test of a hypothesis can be made using an experimental methodology and even then it can never be proven.

Lowland studies

It was felt important that an equivalent species to *R. lanuginosum* should be sampled across lowland Britain. A number of species were considered before finally deciding upon hawthorn (*Crataegus monogyna*) which is commonly planted and managed throughout Britain in hedgerows. Once again the goal was to sample the years growth, but as a deciduous shrubby bush or small tree, it was impossible to sample the total of the seasons growth as with the mosses, since material will be accumulated in wood, bark, buds and roots. Similarly, senescence posed a problem, so sampling took place slightly earlier in late summer, early autumn while the leaves were still green. Samples of hawthorn leaves without any woody material were collected from lowland landscapes, including sites where farm surveys were undertaken.

Although the sampling was undertaken in 1997, there are no reasons to suggest that the atmospheric nitrogen levels would be significantly different from those for the year before, and there is still no contemporary nitrogen deposition figures. Although sources such as animals and vehicles are mobile, the variation in spatial levels at a national scale would be expected to remain constant. When published, the figures are likely to be an average across two to four years.

When analysed the levels in leaves varied from 2 to 3 percent (20 mg g^{-1} to 30 mg g^{-1}), considerably higher than those recorded in the upland mosses and ericaceous shrubs. An analysis of variance (Table 3) showed that there is as much variation within sites as between sites.

Table 3 Analysis of variance for percent nitrogen in hawthorn leaves using the total nitrogen deposition figures as treatment.

Source	DF	SS	MS	F	P
N deposition	17	1.1347	0.0667	1.83	0.106
Error	18	0.6550	0.0364		
Total	35	1.7897			

Pooled Standard Deviation = 0.1908

Figure 7 Relationship between total nitrogen deposition (calculated for forest) and percent nitrogen in hawthorn leaves.

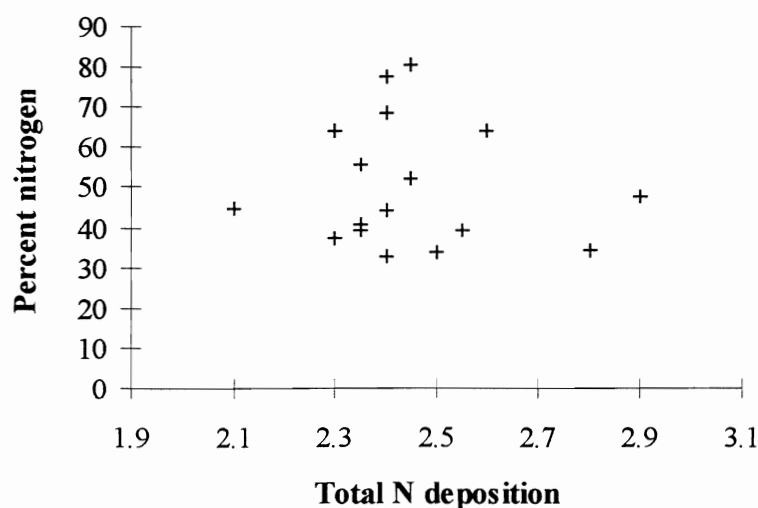


Figure 7 shows there is no correlation between nitrogen level in hawthorn leaves and the national projection of total nitrogen deposition (1992-94). In many ways this is not surprising

as the sources of nitrogen are more diverse and spatially discrete in the lowlands. Typical lowland sources include agricultural management (fertiliser & slurry), gaseous and urinary releases from animals (intense pig and chicken houses especially), vehicles and industrial emissions.

There is only a very weak correlation with area of built up land and roads in and around squares (using data from a 1:250,000 digital map). This is also not surprising as the nitrogen levels are more to do with activities than simple areas.

Discussion

Nitrogen deposition has increased globally approximately fivefold over the last century (Bell, 1994). Although it may be thought of as a benefit to agricultural land, there is no evidence to prove increased yields or reduced fertiliser application because of it. There are several possible reasons for an advantage not being seen which include the chemical form in the atmosphere, the timing of the raised levels (seasonal and daily) and confounding factors from other pollutants (eg sulphur dioxide). There is evidence of a increased nitrogen having a detrimental effect on our semi-natural vegetation.

However, the interpretation of the consequences of increased atmospheric nitrogen is difficult for a number of reasons. Deposition occurs in different chemical forms in different places at different times so there is no single homogeneous consequence. Even with the division of chemical forms into NO_x (predominantly NO_2) and NH_y (predominantly NH_3), the interpretation is not made any easier. The national totals of the oxidised and reduced forms of nitrogen deposition are considered to be approximately equal, their spatial distributions vary independently, depending on the location of sources and methods of deposition. In uplands nitrogen deposition is enhanced by deposition of cloud water.

There are different levels of confidence in the estimates of levels across the country and for different chemical forms. In general, NO_x values are considered more reliable than NH_y and the density of sample sites varies with landscape type. Even so, it is unexpected that one of the surveys shows a reasonable correlation (uplands) while the other shows none (lowlands).

We are, of course, making an assumption that vegetation will reflect its ambient nitrogen environment in the percent nitrogen it contains. For the mosses and *R. lanuginosum* in particular this appears to be true; for the higher plants this may not be the case, and our data is not robust enough to make a definite statement. Not only do we have only part of the season's growth in the hawthorn sample, but the deposition values with which we are comparing are probably more variable in space.

In the uplands, deposition is likely to be more spatially even and much more dependent upon climate (rainfall and cloud); these factors are likely to be normally deterministic in plant distribution. Upland sources of nitrogen include roads, industry and animals; the former carries fewer cars than the lowlands (and is spatially limited), there are fewer industrial sites (mainly power stations) and the density of animals is lower. The predominant deposition is likely to be from the atmosphere generated by remote sources.

In the lowlands there are a large number of small, medium and large sources scattered around the landscape. Some, such as power stations, may use tall stacks to disperse the pollutants and reduce the local effects, but others, such as vehicles, do not. Consequently the ambient nitrogen environment surrounding a hedgerow will depend upon immediate local sources far more than upland sites.

Important factors are likely to be agricultural management in the fields (fertiliser and livestock), traffic density, intensive agricultural units (pigs, poultry and slurry) and industry. To make supported statements about the causes and consequences of nitrogen deposition in the lowlands would require intensive site studies over an extended period.

Figure 8 Variation in mean percent nitrogen of hawthorn sampled in autumn 1996

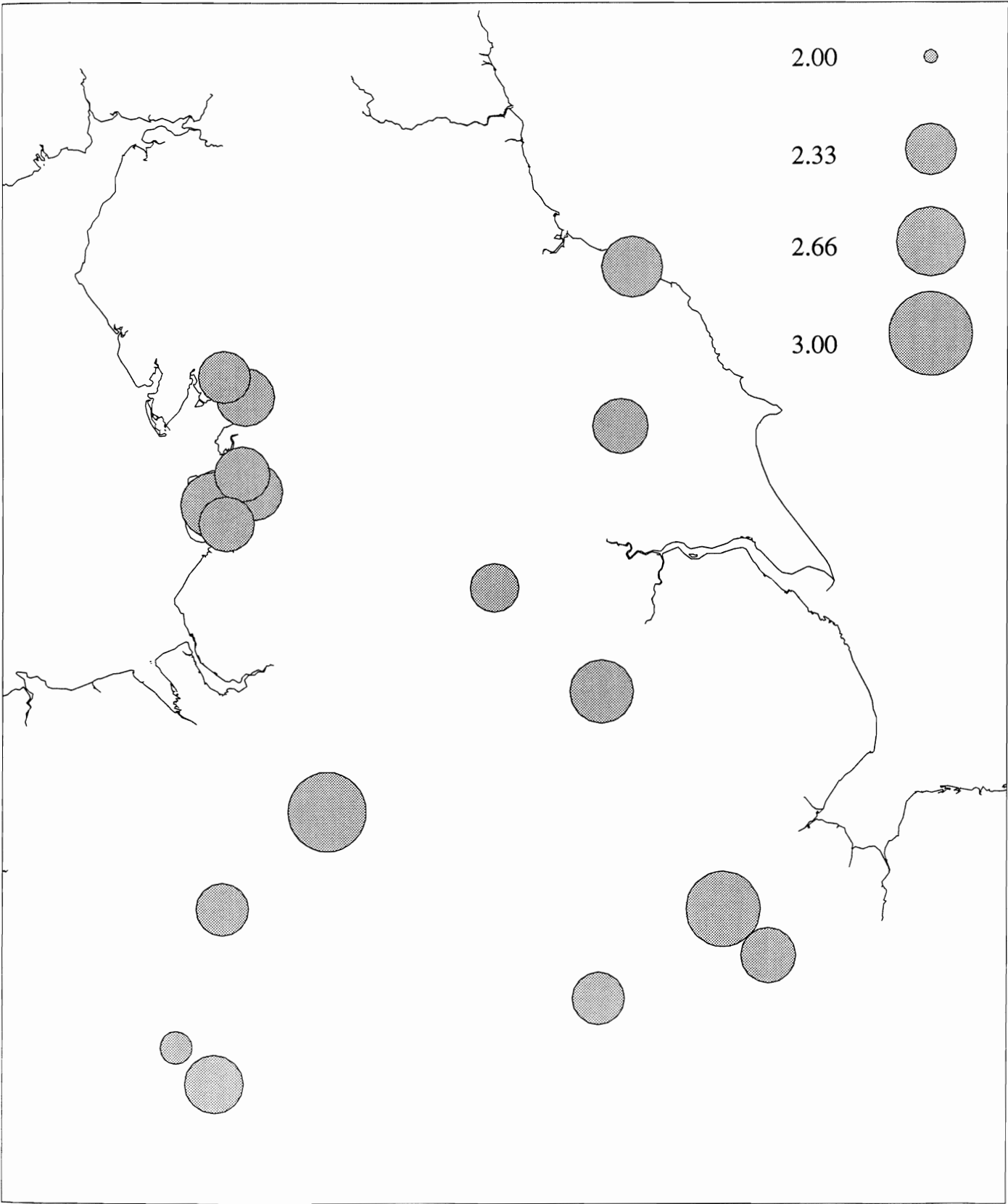
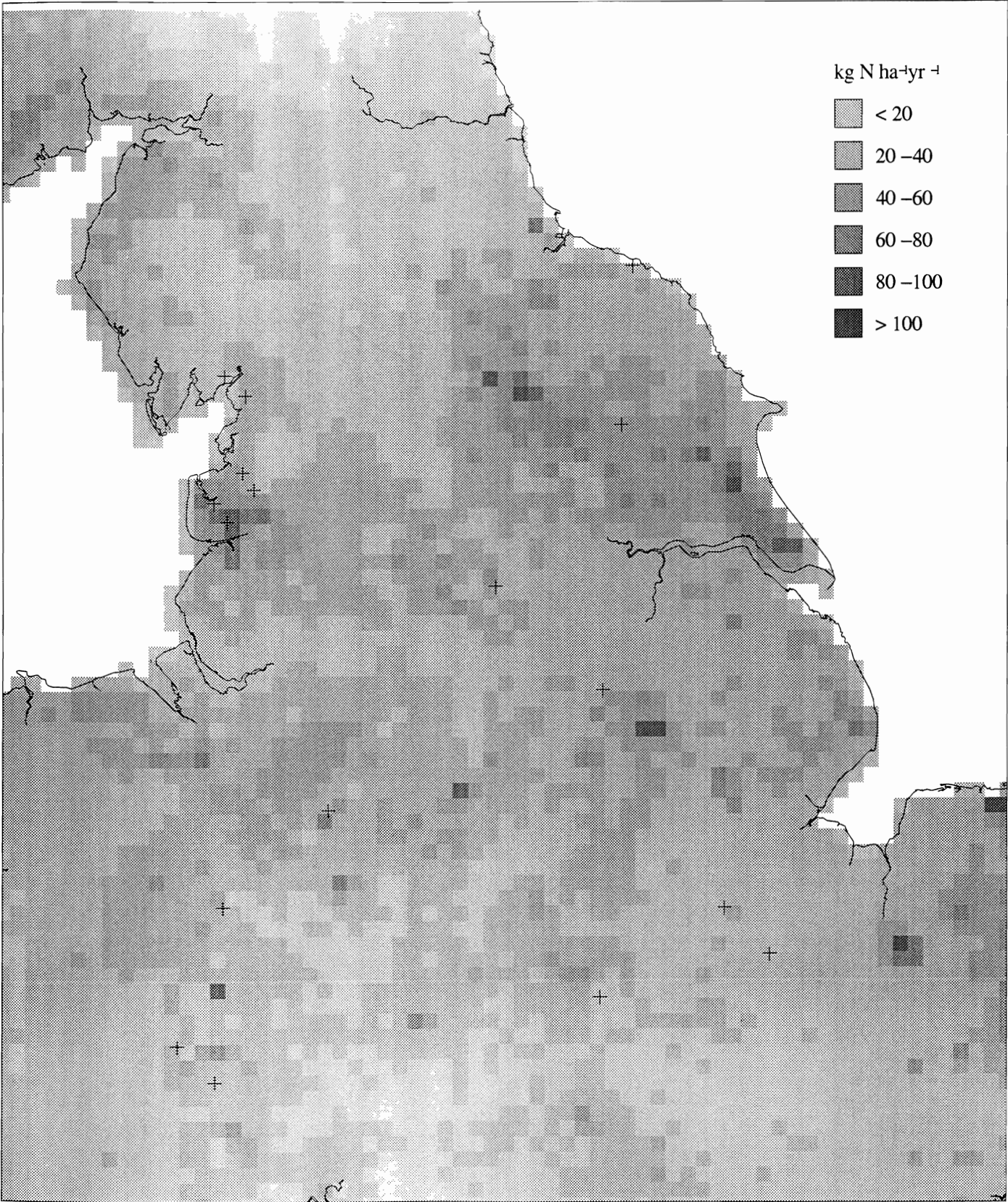


Figure 9 Total nitrogen deposition surface for forestry 1992-94



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The effects of raised productivity on the biodiversity of fields and field boundaries in managed grasslands in GB

Introduction

Results published in the Countryside Survey main report and more recently as part of the EcoFact project showed that between 1978 and 1990 species richness declined significantly in field plots located in unimproved, infertile grasslands. Quality evaluation analyses also highlighted the fact that in 1990 indicator species of unimproved acidic and mesotrophic grasslands (list from English Nature) occupied a higher proportion of boundary plots than field plots in the arable and pastoral landscapes. Although the highest probability of occurrence of both indicator groups was in streamside plots overall, considerable importance attaches to the extent to which agricultural grassland improvement affects the species richness and composition of field boundaries versus field edges. If improvement has an equivalent and negative effect on species richness across field and boundary then in improved or semi-improved situations boundaries will not function as refugia. However even if a refuge function is demonstrated regarding species richness we must still ask whether the identity of the species in the boundary are the same as those that would be in the field given favourable conditions. The study therefore aimed to answer the following questions by sampling from a range of grasslands at various levels of improvement on a latitude gradient

- 1) Is the flow of nitrogen in the vegetation of the field interior the same as that in the boundary ?
- 2) What is the relationship between field species richness and site productivity ?
- 3) What is the relationship between boundary species richness and site productivity ?
- 4) Do the relationships in 1) and 2) differ along a gradient of grassland productivity ?
- 5) Since the species in boundary and field may be different how does species compositional similarity between field and boundary change along a gradient of productivity ?
- 6) Can the relationships in 1) to 4) be demonstrated at the landscape level using Countryside Survey data ?

Methods

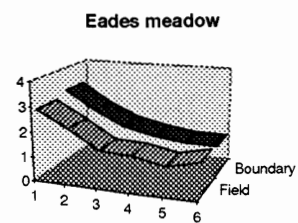
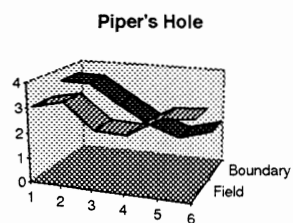
Sampling

Ten managed grassland sites were selected at five locations from Cumbria in the north of England, to Okehampton, Devon in the south west. Levels of agricultural improvement differed across the sampled sites (table 1). Vegetation was recorded and standing crop sampled from three sets of paired boundary and field plots located randomly around each field on six occasions from early May to mid-September 1996. Boundary plots measured 4m along the site edge by 1m into the site. Field plots measured 2m x 2m and were placed 5m from the boundary plot moving perpendicularly into the centre of the site.

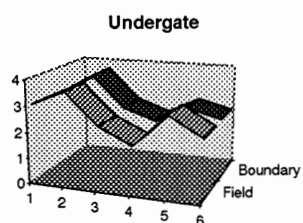
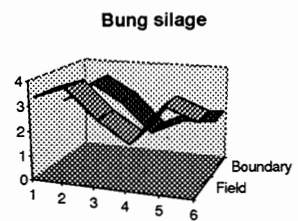
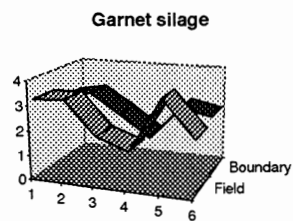
Within each plot all plant species were recorded and cover estimates given to the nearest 5%. Standing crop sampling involved cutting and recording fresh weight in 1m square of every field and boundary plot. A sub-sample was taken, weighed fresh and weighed again after drying for 24hrs at 70°C. Nitrogen content was assayed

Figure 1. Change in N% dry weight through the growing season.

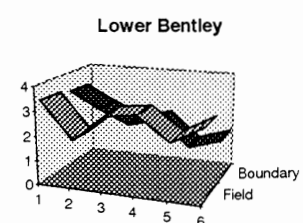
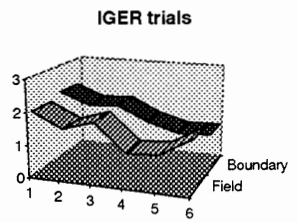
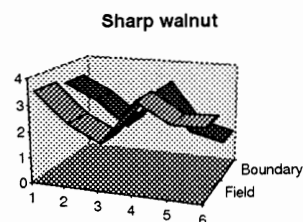
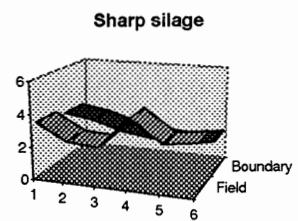
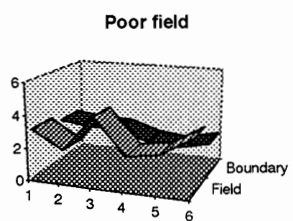
Unimproved grasslands



Semi-improved grasslands



Improved grasslands



by acid digestion and values used to calculate mean N production in the herbage in each field at each sampling occasion.

Sampling through the growing season allowed us to track the changing productivity of each sward. However because fields were cut at least once we could not directly contrast standing crop at each point in time or cumulatively through time between field and boundary. Therefore we arrived at a measure of total field productivity by taking the mean gram N per m² for those periods where the field had reached a standing crop maximum just before harvesting. For silage fields cut more than once we summed the two values available just before each cut. For hay fields we used single values for the occasion before the hay cut.

Analysis

Results are presented graphically. Statistical analyses of association were carried out using the non-parametric Spearman rank correlation. Differences in the relationship between field species richness versus site productivity and boundary species richness versus site productivity were investigated by a t test of difference in slopes (Zar, 1984).

Table 1. Fields selected for vegetation and crop mass sampling.

Site management is indicated by C for silage or hay cut and F for fertiliser application (mineral or FYM). Shaded cells indicate the period over which the field was grazed. Note however, that all sites were grazed close up until the first sampling occasion. Level of improvement are arbitrarily described based upon productivity and species composition.

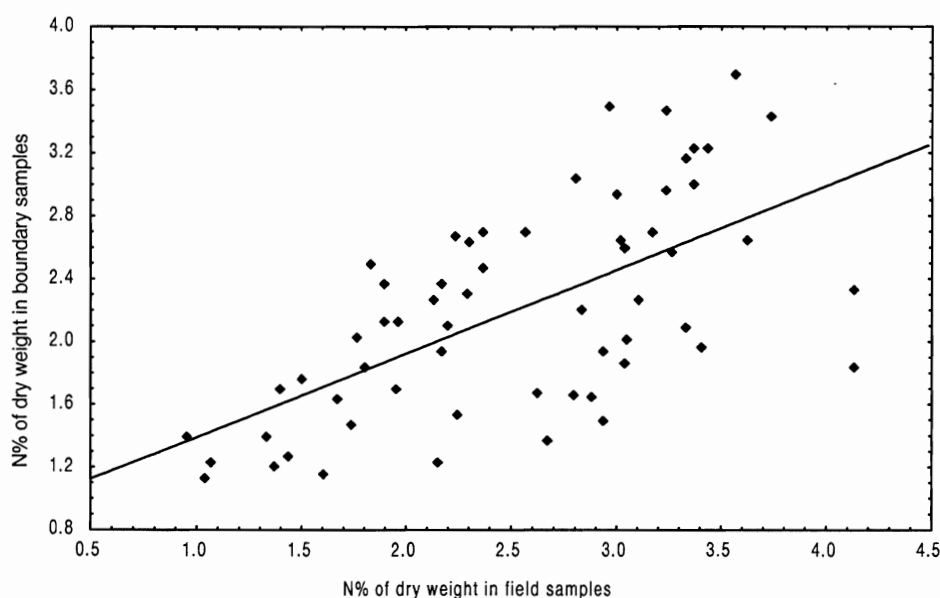
Sampling schedule							mean cum gram N per m ²	Level of improvement
Site name	Northing	1	2	3	4	5	6	
Eades meadow							C	6.39
IGER trials								6.88
Piper's Hole		F				C		8.28
Bung silage		F		C		F		10.3
Garnet silage		F		C		F		10.42
Undergate		F		C				13.03
Lower Bentley		F	C		C/F			15.93
Poor field			C/F		C/F	F		17.76
Sharp silage			C/F			C/F		22.96
Sharp walnut			C/F			C/F		29.56

Results

Nitrogen flow through the season in field and boundary

For all sites, change in proportion of N in the standing crop through the growing season, appeared to follow very similar patterns in the vegetation of field and boundary (Figure 1). However in some fields, for example Bung silage, Garnet silage and Poor field, although patterns of change are similar, trends are dampened somewhat in the boundary plots. The overall positive relationship between field and boundary vegetation is also indicated in figure 2.

Figure 2. N herbage content in field versus boundary. Mean values for ten fields at six times through the growing season. Spearman $R = 0.64$, $p < 0.0001$.



Relationship between species richness and field productivity

Significant negative associations were detected between species richness in both field and boundary plots and productivity. However the slopes of the relationships differ between boundary and field (figure 3) such that field species richness declines at a faster rate than boundary with increase in field productivity. Thus the boundary plots appear to demonstrate a 'refuge' effect. This is explored by plotting values for the difference between field and boundary species richness in paired plots, against field species richness and productivity (figure 4). Note that positive values of field minus boundary richness obviously indicate a richer field vegetation than boundary vegetation and that this applies only at high field species richness and low productivity. Therefore at the lowest productivity field richness surpasses that of the boundary whereas the reverse applies at high field productivity.

Relationship between species compositional similarity between field and boundary and site productivity

Although much variability in similarity coefficient is apparent within sites (figure 5) there is a significant negative association with the highest similarity between field and boundary present at the lowest site productivity. However this

association is only statistically significant when two outlying scores for Eades Meadow are excluded. The implication is that in places within this extraordinarily species rich site there are marked differences in species compoition between field and boundary. We return to this important aspect of the results below.

Figure 3. Relationship between field and boundary species richness along a site productivity gradient. Species richness values taken from all site plots at the time of maximum standing crop before harvest. Field richness versus Productivity; Spearman $R = -0.80$, $p < 0.001$. Boundary versus Productivity; Spearman $R = -0.064$, $p < 0.001$. Slopes significantly different at $p < 0.05$, $t = 2.53$.

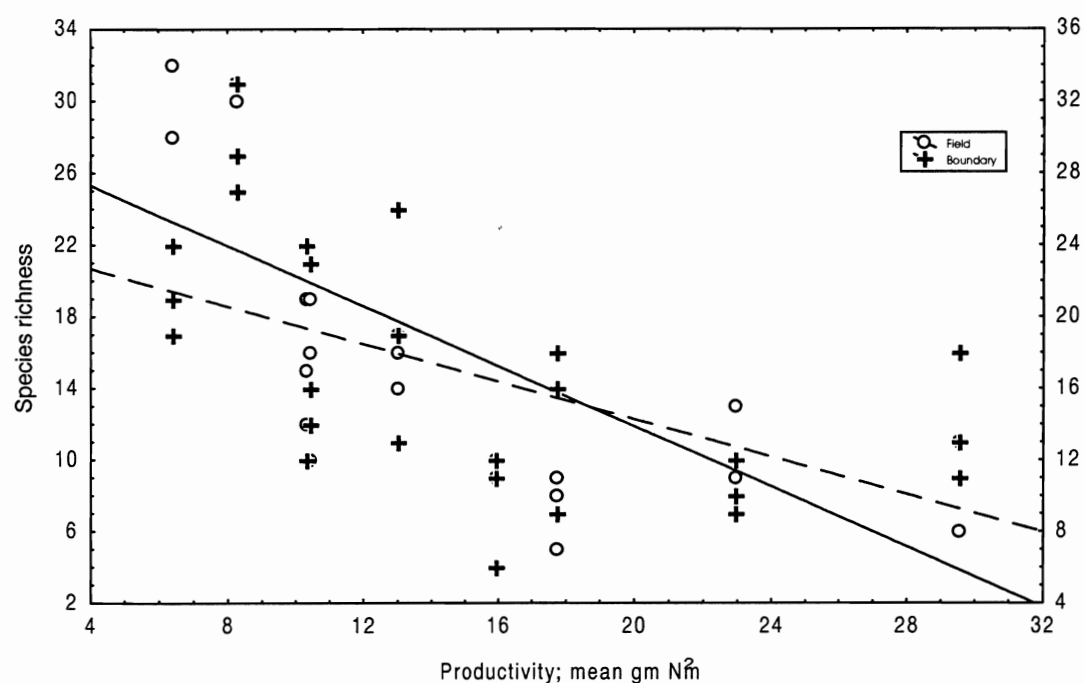


Figure 4. Difference in species richness between paired field and boundary plots along gradients of productivity and species richness.

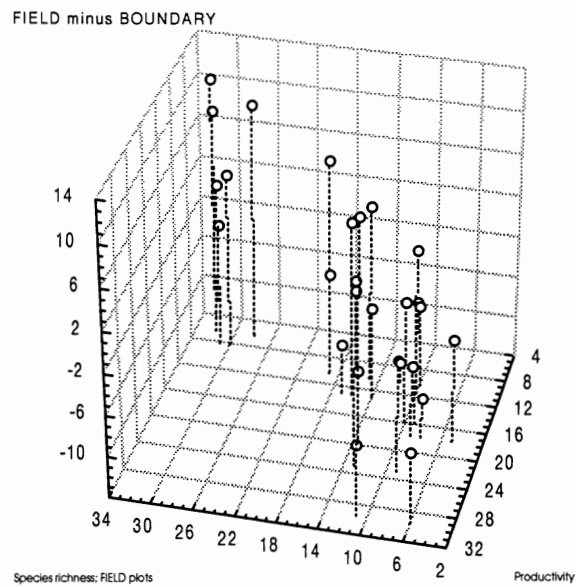
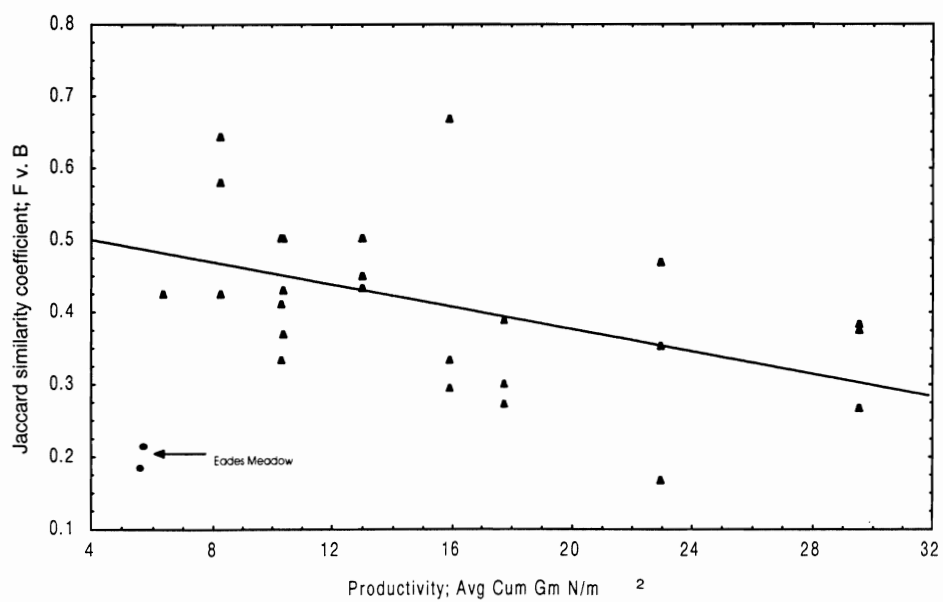


Figure 5. Species compositional similarity between paired field and boundary plots at maximum standing crop along a gradient of site productivity. Spearman $R = -0.51$, $p < 0.01$.



Relationships between boundary and field plots in Countryside Survey data

In order to scale up our investigation of the relationships found above we first needed to derive a surrogate measure with which to define a productivity gradient. Fortunately there is increasing independent verification of the ability of Ellenberg fertility scores to successfully express differences in productivity through space and time (Duckworth, this report, Hill & Carey, 1997, van der Maarel, 1993, Thompson et al, 1993, Bekker et al, 1997) and we indeed found a positive correlation (Spearman $R = 0.88$, $p < 0.01$) between site productivity and a site based Ellenberg score weighted by species frequency across all samples taken through the season.

To explore boundary and field plot relationships we defined a subset of CS1990 data as all paired sets of boundary (B) and field (X) plots where the vegetation of the X plot was classified as one of the three aggregate vegetation classes 2 (tall grasslands), 3 (fertile grasslands) and 4 (infertile grasslands).

Figures 6 and 7 indicate that the 'refuge' effect is also demonstrable within the much larger GB wide CS1990 dataset. Although correlations are significant there is a very substantial amount of variation about the linear regression line; more for the relationship between species compositional similarity than for difference in species richness.

Figure 6. Relationship between field plot productivity and difference in species richness between paired field and boundary plots. Spearman $R = -0.36$, $p < 0.0001$, $r^2 = 0.13$.

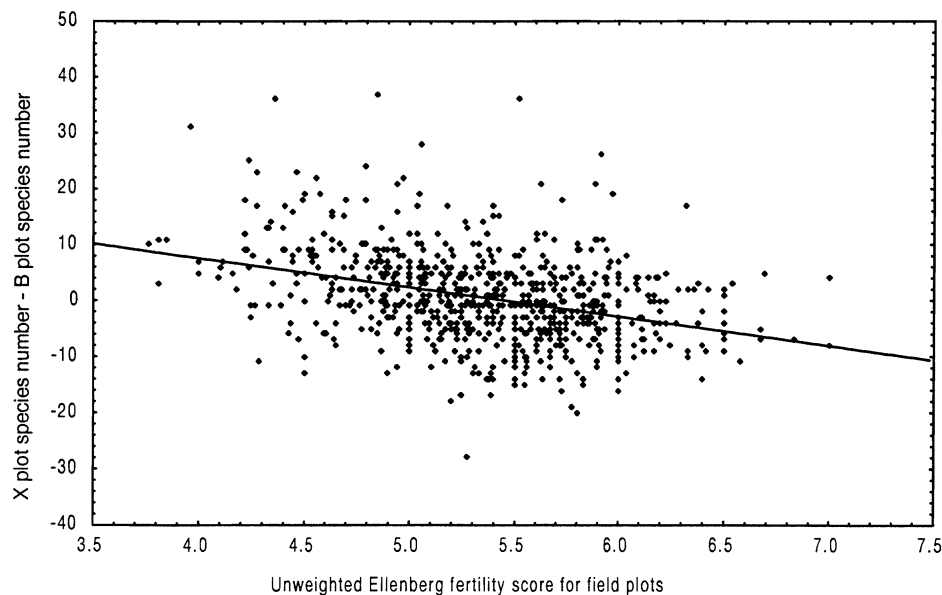
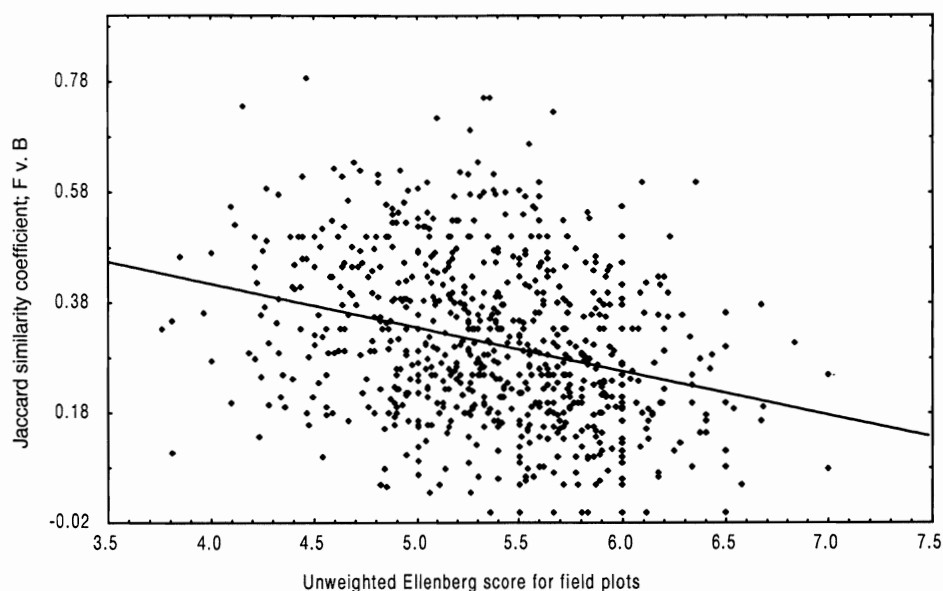


Figure 7. Relationship between field plot productivity and species composition similarity between paired field and boundary plots. Spearman $R = -0.30$, $p < 0.0001$, $r^2 = 0.09$.



Discussion

Species richness of both boundary and field drop as field productivity and therefore the intensity of agricultural management increases. There is however a smaller drop in boundary species for each incremental increase in productivity than within the field proper. This in essence demonstrates the capacity of field boundaries to act as a refuge of species richness.

Examination of change in similarity of species composition between boundary and field also provides evidence that the sheltered assemblages in the boundary partly comprise those that are present in the field at low productivity. However two of the paired sets of plots from the very species rich Eades Meadow NNR had outlying values with very low similarity coefficients between boundary and field even though productivity was the lowest of all sites and field species richness was high. Examination of the relevant species data shows that locally high beta-diversity characterises the transitions from field to boundary and from an unusually rich meadow assemblage to a still rich boundary vegetation where species characteristic of less disturbed, partially shaded and probably more fertile conditions coexist with relict woodland species such as *Hyacinthoides non-scripta*, *Primula veris* and *Colchicum autumnale*. The greater floristic differentiation between boundary and field at low productivity is also emphasised by allocation of plots in Eades Meadow to different CVS plot classes and to different NVC units (appendix A). Appendix A results further highlight the importance of boundary conditions pertaining to each site since at Piper's Hole well maintained stone walls on level ground are associated with the effects of field management being felt right up against the wall. Thus the boundary and field plots are hardly differentiated on floristic grounds. These two examples suggest that any general relationship between boundary and field species composition

is likely to be profoundly shaped by local site conditions, for example differential effects of shade, slope, ground wetness and soil reaction as well as differences in fertility and disturbance related to the moderated or even nil impact of field management on the boundary. Moreover net effects on beta-diversity are likely to be more sharply expressed in a low productivity situation.

Using CS19990 data we find that both relationships are still detectable at the landscape scale. This is an important result as it proves that on average boundaries do function as partial refugia but crucially are themselves still negatively affected by increasing intensity of management, just to a lesser extent than the field interior. Also the very large amount of variation not explained in CS1990 data shows how important local and site specific conditions are in modifying these general trends.

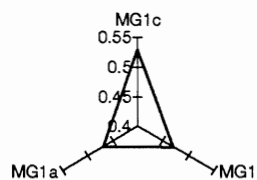
Conclusions

- Field and boundary species richness both declined with increasing field productivity in a sample of ten grassland fields from throughout GB.
- Boundary species richness declined less rapidly however, and the difference in rate of species reduction between field and boundary is detectable as a 'refuge' effect.
- The same refuge effect is demonstrable for species compositional similarity between boundary and field along a productivity gradient.
- Both relationships are found at the wider landscape scale but a great deal of variation remains unexplained. This variation highlights the probable importance of local site conditions.

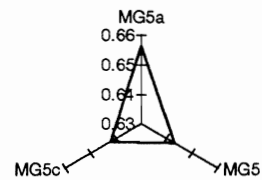
Appendix A

Unimproved grasslands

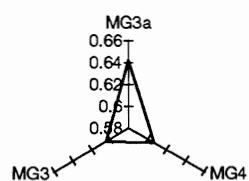
Eades Meadow; boundaries



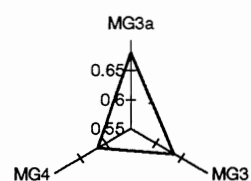
Eades Meadow; fields



Piper's Hole; boundaries

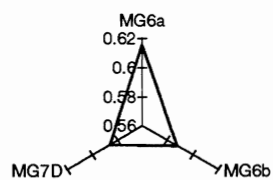


Piper's Hole; fields

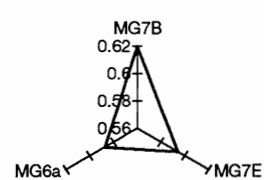


Semi-improved grasslands

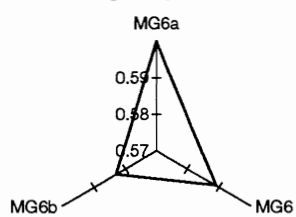
Garnet silage; boundaries



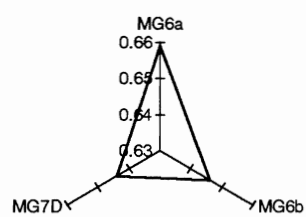
Garnet silage; fields



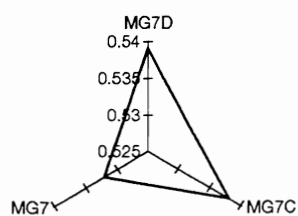
Bung silage; boundaries



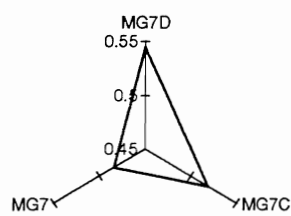
Bung silage; fields



Undergate; boundaries



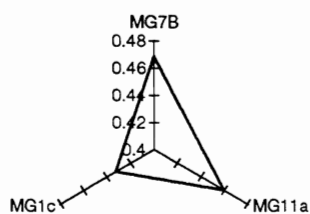
Undergate; fields



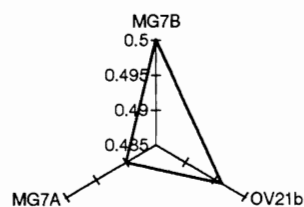
Appendix A continued.

Improved grasslands

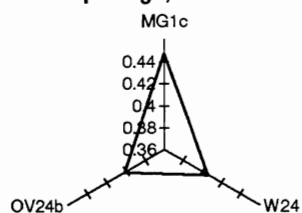
Poor field; boundaries



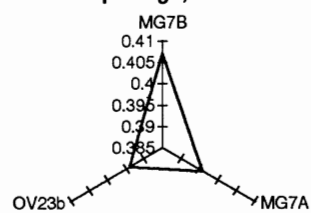
Poor field; fields



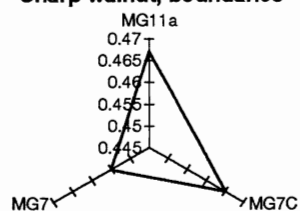
Sharp silage; boundaries



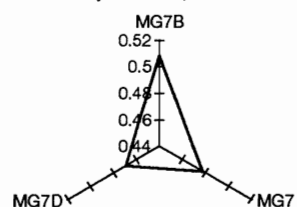
Sharp silage; fields



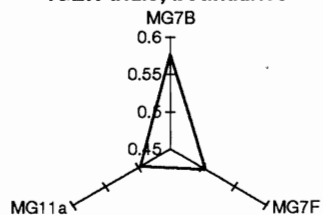
Sharp walnut; boundaries



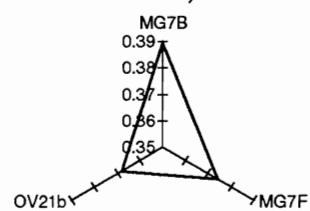
Sharp walnut; fields



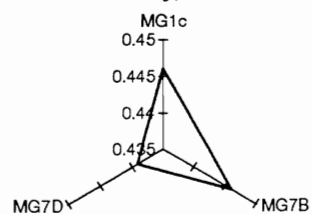
IGER trials; boundaries



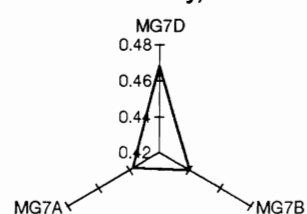
IGER trials; fields



Lower Bentley; boundaries



Lower Bentley; fields





A Functional Interpretation of Landscape Scale Vegetation Changes Detected over a 12 Year Period in the British Isles.

Introduction.

Large changes in the extent and composition of many vegetation types throughout Britain have occurred over the last few decades (Ratcliffe 1984, Way *et al.* 1988, Hopkins 1988, Rich & Woodruff 1996) as well as in the management they receive (Stoate 1996, Ilbery 1992, Church & Lewis 1977). Changes in species composition and abundance and in species number within most extensive vegetation types across Britain were detected over the 12 years up to 1990 (Barr *et al.* 1993). Such consistent and widespread change throughout the British Isles has prompted fears for the loss of characteristic species and vegetations (Marren 1996, Way *et al.* 1988, Green 1984). There is much evidence suggesting that land use processes are the dominant force controlling the distribution and abundance of species in the British landscape (Thompson 1994, Wells & Sheail 1988, Hodgson 1986, Ratcliffe 1984) and a link can therefore be suggested between these processes and the vegetation changes observed.

This study combined functional data on plant species traits with vegetation composition data from over 2000 permanent plots situated throughout Britain and surveyed in 1978 and 1990. The functional compositions of plots grouped by landscape type, landscape unit (fields, hedgerows, road verges etc.) and vegetation type were used to ask the following questions

- Are changes in the composition of vegetation types associated with consistent shifts in the functional characteristics of the vegetation?
- What processes of change can be inferred to be driving these functional shifts?
- How do these processes relate to changes in land management practises?
- How do processes of change differ between landscape types, landscape units and vegetation types?

Materials and Methods.

This study used species composition data from over 2000 permanent vegetation plots

recorded in 1978 and 1990 as part of the Countryside Survey by the Institute of Terrestrial Ecology. The plots were located in 256 1km National Grid squares throughout England, Scotland and Wales and stratified to give equal numbers in each of the ITE land classes (see Barr *et al.* 1993). All vascular plants plus restricted numbers of Bryophytes and lichens present in the plots were recorded (Barr 1990).

In order to analyse plots with broadly similar vegetation, land use, management and terrain, a 3-tier stratification was used to produce groups of plots for analysis. Plots were split into 8 statistically derived vegetation 'aggregate classes' based on their vegetation composition (see for details). The 4 ITE landscape types (aggregates of the land classes) were then used to create 32 possible groupings. These were then stratified according to the 4 types of vegetation plot that were recorded (open vegetation 'main' plots that included no boundary features, road verge plots, hedgerow plots and streamside plots) giving 128 possible groupings. Higher level aggregates of these groupings were also analysed.

Not all 128 groupings were viable, there were no 'crop vegetation' hedgerow plots in the upland landscape. Any grouping that included at least 10% of the particular plot type in that landscape type was analysed.

Due to the statistics used in deriving the 8 vegetation aggregate classes, a plot could be assigned to different classes based on the 1978 and 1990 records. The 8 aggregate classes for analysis were therefore derived in 3 separate ways, the 3 analyses emphasising different features of the vegetation change data. In the first analysis paired plot records for 1978 and 1990 were assigned to an aggregate class based on the 1978 classification for the plot. This looked at change in a given group of plots but included gross vegetation shifts due to changed land use such as upland afforestation. The second analysis only used paired 1978 and 1990 data from plots that were classified into the same aggregate class in both 1978 and 1990. This reduced the data available by about 40% but excluded gross management changes. The third analysis split up paired plot data and classified 1978 and 1990 plot records separately according to the original statistical derivation of the aggregate classes. This used all the data, excluded gross management changes but did not look at change in a set group of plots from a

common starting point.

For each group of plots, the proportional change in the number of records for each species that occurred in more than 2 plots in at least 1 year, and excluding species that were not present in 1978, was calculated. This profile of change in species presence was correlated with 25 plant functional traits obtained from databases at the Unit of Comparative Plant Ecology in Sheffield. Spearman non-parametric correlation coefficients were calculated for each group of plots (SPSS 6.0 for Windows), between the set of species proportional changes and the values of each functional trait for the species.

The functional traits used relate in a predictive manner to aspects of species ecology related to land management and land use such as site nutrient status and disturbance regimes. 3 types of traits of increasing reliability as predictors of species ecology were used; firstly those derived from species distribution in different habitats and obtained from a separate vegetation survey of Central England (distribution traits); secondly, easily measured plant attributes that correlate with key elements of species ecology ('soft' traits) and lastly those that are actually measures of key ecological attributes ('hard' traits). Table 1 lists the traits used and how they relate to species' ecology and ecological processes.

This analysis generated sets of significant ($P \leq 0.05$) correlations for each group of plots. These were interpreted as indicative of different processes of change, taking into account the significance level of the correlations, the types of the significantly correlated traits and the consistency with which all significant correlations indicated the same process.

The processes of change inferred in plot groupings were summarised under 3 main processes. These were eutrophication, dereliction and increased disturbance. The functional traits used also relate to other vegetation processes that are less fully understood functionally. Not all sets of significant correlations are thus fully interpretable.

Results.

Cases Analysed.

42 groups of plots were eligible for analysis in each style of analysis out of the possible 128.

41 higher level groupings of plots were also analysed in each case. In the analysis using a reduced dataset only 30 groups plus 40 higher level groupings were analysed due to inadequate numbers of plots available.

Processes Inferred.

In the first analysis 12 out of 42 plot groups and 17 out of the 41 higher level groupings showed sets of significant correlations indicative of processes of change related to land use. The average number of significant correlations for these groups was 8.1 (s.e. 0.53). The average number of significant correlations for other plot groupings was 2.7 (s.e. 0.25). In the second analysis using a reduced dataset 6 out of 30 plot groups and 9 out of the 35 higher level groupings showed sets of significant correlations indicative of processes of change related to land use. The average number of significant correlations for these groups was 5.3 (s.e. 0.47). The average number of significant correlations for other plot groupings was 1.7 (s.e. 0.22). In the third analysis 12 out of 42 plot groups and 8 out of the 35 higher level groupings showed sets of significant correlations indicative of processes of change related to land use. The average number of significant correlations for these groups was 5.2 (s.e. 0.55). The average number of significant correlations for other plot groupings was 2.6 (s.e. 0.25).

In the plot groupings with inferred processes of change in the first analysis, 9, 20 and 4 groups of plots indicated eutrophication, dereliction and increased disturbance respectively. More than one process may be inferred for each plot grouping, eutrophication often occurs in tandem with other processes. In the second analysis with reduces data, the figures were 12, 5 and 3 groups of plots indicating eutrophication, dereliction and increased disturbance respectively.. The third analysis gave figures of 12, 6 and 5 plot groups for the same three processes.

No plot groupings showed sets of significant correlations indicative of reduced site nutrient status, the opposite of eutrophication and a process not thought to be widespread in British landscapes.

Eutrophication occurs in all plot types as does dereliction. However, road verges and

hedgerows do not suffer increased disturbance, a process mainly confined to the 'main' open plots.

Discussion.

Following the Divergent Fates of Plots from a Common Starting Point.

The first analysis looks at change within a set group of plots that may have been subject to abrupt changes (Bunce & Heal 1984) in land use and management. Thus upland grassland may have been afforested or lowland grassland ploughed for arable usage. More subtle changes between grassland classes may also be detected. These are the changes associated with species loss and shifts of vegetation character that are of interest in this study. Matrices of change in the numbers of plots in different vegetation types are available () and these lend support to conclusions about processes occurring.

All processes inferred from groups of plots having crops type vegetation in 1978 were dereliction. Life length, plant size and competitive ability (*sensu* Grime 197) were all positively associated and association with arable landscape was negatively associated with species increase in crops vegetation. This is in agreement with the 23% of crop plots in the arable landscape and the 30% of crop plots overall that shifted to grassland vegetation between 1978 and 1990 (). More detailed examination of changes to crop type vegetation will be seen in a later analysis.

Plots that were classified as tall grassland in 1978 also suggest dereliction in this analysis. This process occurs in hedgerows across all landscape types, other landscape elements remaining essentially unchanged in terms of the functional traits used in these analyses. Plant size and ability to spread laterally, association with wooded and shady habitats and competitive ability are all positively associated and ruderality (*sensu* Grime 197) and association with more species rich, more regularly managed habitats such as pastures and arable habitats are all negatively associated with species increase. These all suggest that this type of hedgerow vegetation is being less frequently mowed, cut or grazed and is becoming denser and more

overgrown over the period of the survey. 19% of tall grassland vegetation as a whole changed to lowland woods and hedges over the time period indicating this same shift. The % change figure for the hedgerow plots is unknown however.

The only process of change implied in eutrophic grassland is dereliction and this occurs only in lowland linear plot types. 'Main' open vegetation plots represent the most intensively managed grasslands that have remained functionally very similar over the survey period. The same is not true of road verges where dereliction, again maybe due to reduced mowing and cutting may be occurring. Such small areas of grassland in the lowlands have increasingly lost their management due to economies of scale (Morris 1984). The trend is towards competitive, large, long-lived species characteristic of shaded or wooded habitats increasing at the expense ruderal species of more species rich, regularly managed or disturbed habitats. This is in agreement with observed movements of plots from eutrophic grassland vegetation to tall grassland and lowland grassland vegetation types characterised by lower disturbance incidence and less strongly dominated by species of very high fertility habitats. How many of the changing plots are of each plot type is unknown however. A large change (of presumably main plots) to crops vegetation may confuse any processes of change occurring in these plots, this gross land use change associated with increased intensity of management in these areas (Morris 1984) shifting vegetation in functionally different directions to more subtle shifts that may be occurring.

Implied processes of change in lowland grassland vegetation of dereliction, plus eutrophication in the marginal upland landscape, imply reduced management throughout this vegetation type particularly in linear plot types. Eutrophication in the marginal uplands may be linked to the pattern of improvement of grasslands in these areas (Eadie 1984).

In lowland woods and hedges, eutrophication is implied in lowland hedgerows. The only major change between vegetation classes is from lowland woods to tall grassland, a class characterised by fewer species of extremely nutrient poor habitats. It is highly likely that this change is due to drift and drainage of fertilisers from adjacent fields in these landscapes where there was extremely high usage of fertilisers over the survey period.

Increased disturbance of acid woodland streamside plots is implied.

No shifts in upland grasslands were implied.

A combination of eutrophication and increased disturbance is implied in bogs and heaths that lie mainly in the upland landscape. Large changes in the management of upland landscapes have occurred during the survey period, many of which could be linked to this change. Only a small shift of plots to upland grassland occurred although this may be driving the apparent change as no change is implied in other analyses of this vegetation type. Upland grassland is characterised by fewer stress-tolerant species of low-nutrient habitats and may be used more by sheep for grazing, a land use that has changed in intensity greatly over the survey period in some areas. Grazing of moorlands is known to cause a shift to grassland at higher intensities (Hudson 1984, Armstrong & Milne 1995, Gimingham 1995, Nolan *et al.* 1995). Such a shift may also be linked to afforestation and the associated ploughing and fertilisation, a process that has occurred over huge areas over the survey period (Mutch 1984, Tudor & Mackey 1995).

Detecting Subtle Changes in Plots Whose Vegetation Remained Superficially the Same.

This analysis again looked at changes in set groups of plots but excluded plots where abrupt changes in land use and management had shifted the vegetation to one of the other statistically derived vegetation types by 1990. Shifts due to more subtle management processes were also excluded however where they caused shifts between vegetation types that essentially lay adjacent on a continuum of management intensity. This analysis therefore attempted to look at vegetation shifts due to the more gradual and subtle changes in land use that may not be apparent from overall land use and vegetation data (Bunce & Heal 1984).

Implied changes to crops vegetation are changed disturbance regimes in the arable landscape. As the majority of crops vegetation plots are in the arable landscape, this effect is also seen in the vegetation type as a whole. Ruderality, seed weight and plant canopy height are all positively correlated whilst stress tolerance (*sensu* Grime 197), ability to spread laterally, length of life history and life length of soil seed bank are all negatively correlated with species

increase. These changes are due to an increase in a very few large-seeded, tall annuals at the expense of all most species associated with crops vegetation. This is thought to relate to a widespread change from spring to autumn sown cereals in Britain during the survey period. Minimum tillage associated with autumn sown cereals favours large-seeded, autumn germination species without a long-lived seed bank that cannot tolerate deep burial associated with ploughing (McCloskey *et al.* 1996, Chancellor *et al.* 1984). Two of the species increasing the most: *Anisantha sterilis* and *Galium aparine* are the species also found to increase by McCloskey *et al.* (1996) under minimum tillage systems and exhibited significant peaks of autumn germination in the work of Froud-Williams *et al.* (1984). This increase in a few species is associated with the decrease of many broad-leaved weed species, the change in cultivation time changing the weed flora composition (Roberts 1984). Many of the perennial weed species of field margins were also lost as ploughing was taken right up to the hedge base increasingly from the 1970s and herbicide drift encouraged vigorous grass species at the expense of perennial species (Stoate 1996).

The only process indicated in tall grassland was eutrophication in the arable landscape. This is not indicated in the individual plot types, perhaps due to insufficient numbers of plots available. In the arable landscape, such plots presumably lie adjacent to cultivated fields and eutrophication may occur as fertilisers drift from adjacent land. This is not seen in the pastoral landscape.

The same applies to eutrophic grassland where eutrophication is implicated in the arable landscape, presumably for the same reasons.

Lowland grassland also shows indications of eutrophication occurring, but the traits of increasing species also point to dereliction, that is cessation of management or grazing. These two processes are inferred throughout the vegetation type in both the pastoral and marginal upland landscapes wherever significant processes are detected. It is unknown whether the processes occur in the same groups of plots or not. Large increases in fertiliser application over the survey period have occurred (Hopkins and Wainwright 1989) and large changes in management with a large shift from hay to silage production (Hopkins 1988), the areas under

silage production receiving higher inputs of nitrogenous fertilizer (Hopkins *et al.* 1988). This process either didn't occur in the arable landscape or that occurred to such an extent that plot vegetation shifted to another vegetation class.

The implications are for eutrophication of lowland woods and hedge vegetation in arable hedgerows, a process that may be related again to fertiliser drift in a landscape where fertilisers were applied in great and increasing quantities between 1978 and 1990.

In the other three predominantly upland vegetation types, no indications of processes of change were detected. If other analyses do detect changes in these plots, it suggests that change may be limited to those plots where change occurred enough to shift the vegetation to another vegetation type.

Detecting Changes in Plots with Similar Vegetations and Management.

This analysis looks at changes in groups of plots that had similar vegetation compositions in the two years. This analysis assesses whether there has been any overall change in the character of a particular vegetation type across Britain and does not necessarily look at changes within set groups of plots. As plots can shift between vegetation types between the two years, a lack of change in this analysis may be detected if the plots shifting into the vegetation type balance out the plots that leave the vegetation type, leaving the overall character the same.

The same process is implied for crops vegetation as in the previous style of analysis. The species increasing the most is again *Anisantha sterilis* probably due again to the change from spring sown to autumn sown cereals (McCloskey *et al.* 1996). Most species in this vegetation type have decreased over the survey period. Other consistently increasing species are *Tripleurospermum inodorum*, *Capsella bursa-pastoris*, *Viola arvensis* and *Alopecurus myosuroides*. Both *T. inodorum* and *A. myosuroides* exhibit marked peaks of germination in autumn (Froud-Williams *et al.* 1984) in the same way as *Anisantha sterilis* and *Galium aparine*. *C. bursa-pastoris* and *T. inodorum* were found to germinate at any time of the year (Roberts 1964). *T. inodorum* and *A. myosuroides* were also found to increase under herbicide

treatment of winter wheat (Fryer & Chancellor 1970).

Dereliction is inferred for tall grassland plots as in the first analysis, particularly for arable landscape hedgerow plots, and also for road verge plots. Elsewhere, the overall character of the vegetation type did not change.

The overall character of eutrophic grassland remained unchanged across the whole of Britain. This may well be the result of changes in plot classification balancing out overall, particularly as this grassland type appears to lie on a continuum with the other two grassland types.

More is inferred in the way of change in the lowland grassland with different processes being inferred for each landscape type. In the arable landscape, this vegetation type became more disturbed as a whole, which may be due to the fact that nearly 50 % of the plots of this vegetation type had changed to another vegetation type by 1990. The influx of plots being of more disturbed vegetation types.

Eutrophication is implicated in both the pastoral and marginal upland landscapes in this vegetation type. This would occur if grassland were improved over the survey period by additions of fertiliser (Hopkins 1988, Wells & Sheail 1988).

Lowland woods and hedges appear to have become more eutrophic in the arable landscape, particularly in linear plot types (hedges and streamsides) where such an effect could occur as fertilisers drift from adjacent arable fields.

This analysis implies changes occurring in the acid woods vegetation type in the upland landscape. Eutrophication in this landscape may occur due to deposition in these areas of nitrogen oxides filtered out of the atmosphere by coniferous trees. Eutrophication may also be due to addition of fertilisers to upland coniferous plantations.

No changes are inferred in upland grasslands or heaths and bogs vegetation types.

Trait	Ecological correlate	References
Distribution-related traits.		
Affinity with wetland habitats	n/a	

Affinity with skeletal habitats	n/a
Affinity with arable habitats	n/a
Affinity with grazed habitats	n/a
Affinity with cut/mown habitats	n/a
Affinity with traampled habitats	n/a
Affinity with pasture habitats	n/a
Affinity with spoil habitats	n/a
Affinity with wasteland habitats	n/a
Affinity with shaded habitats	n/a
Affinity with woodland habitats	n/a
No of associated species / m2	High species diversity associated with low fertility and infrequent disturbance
Life history length	Short life history associated with frequent disturbance.
Competitiveness (<i>sensu</i> Grime)	Ability to dominate in productive undisturbed systems
Stress tolerance (<i>sensu</i> Grime)	Ability to persist at low levels of productivity and low disturbance.
Ruderality (<i>sensu</i> Grime)	Ability to persist in systems subjected to frequent disturbance.
Time of flowering start (month of year)	Competitiveness linked with extended period of vegetative growth prior to flowering.
Length of flowering period (months)	
Seedbank (longevity)	
Specific leaf area	Linked to ability to achieve high growth rates.
Species canopy height	Ability to successfully compete for light in productive conditions.
Species lateral spread	Ability to monopolise space under conditions conducive to high growth rates.

Seed weight		
Leaf nitrogen content	Preferred site nutrient status.	
Leaf phosphorous content	Preferred site nutrient status.	

Table 1. Species functional traits used in analysing changes in species composition of resurveyed plots.

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Functional Analysis of Countryside Survey Vegetation Data.

Tables 1 to 3 show the results of functional analyses carried out on Countryside Survey vegetation plot data split by ecotope group, landscape type and plot type, the data being split differently in each table. As well as results at the level of single plot types in single landscape types in each ecotope group, data for groupings of plots at higher levels are also included. This overcomes problems associated with the small samples of plots in certain groupings. It also validates some of the observed functional shifts seen at fine divisions by showing the same processes in higher aggregations of plots. This may strengthen evidence of a consistent functional shift in species characteristics.

Definition of the 3 Analyses.

Detecting changes in plots with similar vegetation and management.

Table 1 contains the results from an analysis using 1978 data from plots that were classified as belonging to the particular ecotope group in 1978 compared to 1990 data from plots that were classified as belonging to the same particular ecotope group in 1990. Thus, the exact plots in 1978 and 1990 may not be the same. This analysis will be referred to as 'simple' analysis.

Detecting subtle changes in plots whose vegetation remained superficially the same.

Table 2 contains the results from an analysis of data from only those individual plots that were classified as belonging to the same ecotope group in 1990 as in 1978. This analysis will be referred to as 'stay the same' analysis.

Following the divergent fate of plots from a common starting point.

Table 3 contains the results from an analysis of data, the plot data from both 1978 and 1990 grouped according to the ecotope group into which the plot was classified in 1978, irrespective of which ecotope group they belonged to in 1990. This analysis will be referred to

as '1978-based' analysis.

Results

The figures in tables 1 to 3 are the numbers of significant correlations (significant at $P=0.05$ or less) between the proportional change in species abundance (based on the number of plots in which it occurred) and the values of various traits for the species. The numbers in bold indicate those situations where the set of correlations suggests a consistent process of change affecting the group of plots between the 2 dates. Other values represent sets of correlations of uncertain ecological significance.

The magnitude of the number of correlations cannot be taken as an indicator of processes of change within the plots. The traits include those derived from plant species distributions (based on surveys of vegetation in Central England), through traits of plant morphology derived from floras (e.g. plant height) to reliable, traits predictive of species' ecology (e.g. leaf mineral nutrient contents relate to the nutrient status of the species' preferred habitats). Thus, a large number of correlations with the less reliable distribution data may be less indicative of change than a smaller number of correlations with hard, predictive traits. Furthermore, in situations where processes of change have affected the plots in more than one direction, a confused set of significant correlations may give an apparently self-contradictory picture. The bold figures are thus based on interpretation of the raw correlation data. brief commentary is also provided.

Discussion and Interpretation of Results.

'Simple' Analysis.

Crops ecotope group. Viable groups of plots occur in only 2 landscapes and significant changes are seen only in the arable landscape. The changes indicate increases in disturbance in linear as well as main plots. The consistent increases in species' seed weight and plant canopy height are associated with large-seeded ruderals that germinate in the autumn. The change is

thought to be linked to a change from spring to autumn sown crops. No such changes occurred in the pastoral landscape.

Tall grassland again only shows change in the arable landscape. When all landscape types are analysed together, these changes are masked by the lack of change or contradictory changes in other landscape types. Changes in both road verges and hedges in the arable landscape are quite strongly indicative of dereliction as both plot types become less ruderal and more dominated by competitive species or species with extensive canopies that undergo an extended period of growth before flowering. This pattern is not seen in streamside plots.

Eutrophic grassland shows very little in the way of consistent change. The ecotope group name suggests that this vegetation is already at the intensively managed end of the productivity gradient within vegetation and therefore unlikely to be further changed. The only change is seen in the marginal uplands where intensification may be less advanced, however the correlations are not strongly indicative of any particular process.

Lowland grassland shows many significant changes, particularly in streamside and main plots. The data also provide a good example of the efficacy of using a wide variety of plant traits. In several cases subsets of the data indicate the same changes as higher groupings but via significant changes in different variables; the 'belt and braces' philosophy of using a variety of 'soft' predictive traits. The changes observed also differ between landscape types.

Results from arable landscape streamside, main plots and all plots indicate increases in disturbance as they all show increases in species richness and in species of disturbed habitats at the expense of species of more closed habitats. Smaller seeded species also seem to be increasing.

In the marginal upland landscape the changes to streamside and main plots seem to be in the direction of eutrophication. Both groupings show correlations that may indicate this process but via different sets of traits. The process is masked at the whole landscape level by many hedgerow and road verge plots that do not show the same changes.

The pastoral landscape shows different processes of change, both streamside and road verge

plots having correlations that suggest processes of dereliction or eutrophication. The results for the whole landscape type strongly suggest the process of eutrophication. This is a very abundant and widespread ecotope group in the pastoral landscape and changes may be going on in more than one direction within the groups, particularly the main plots. However, eutrophication throughout the landscape type is strongly indicated.

Not surprisingly, no indication of processes of change is obvious at the all landscape types level due to the variety of processes identified in individual landscapes.

Lowland woods and hedges. Here again change is suggested in the arable landscape and not in the pastoral landscape. The correlations for the arable landscape suggest that eutrophication may be occurring, a process indicated for the whole ecotope group throughout all landscape types.

Acid woods is a relatively limited type but still shows processes of change affecting vegetation between 1978 and 1990. Change is seen in the upland landscape where two reliable correlations indicate increases in species of nutrient rich habitats. When all upland plots are analysed the suggestion of eutrophication is much greater. Large, competitive species of nutrient rich habitats are increasing at the expense of stress tolerant species of species rich habitats. There is also an indication of dereliction, a process indicated for all the streamside plots of this ecotope group.

Upland grassland shows little evidence of change between 1978 and 1990.

Bogs and heaths again showed no overwhelming indications of change in the functional make-up of the vegetation. The large number of correlations in the marginal upland landscape are largely distribution related traits of little indicative value in this case.

Summary.

‘Simple’ analysis. *Detecting changes in plots with similar vegetation and management.*

Changed disturbance in **crops** group in arable landscape.

Dereliction of road verges and hedgerows in **tall grassland** in arable landscape.

Increased disturbance in **lowland grassland** in arable landscape.

Eutrophication of streamside and main plots in **lowland grassland** in marginal uplands.

Eutrophication of **lowland grassland** in pastoral landscape.

Eutrophication of **lowland woods and hedges** especially in arable landscape.

Eutrophication of **acid woods** in the upland landscape.

'Stay the same' analysis

This analysis includes fewer plots than either of the other 2 analyses and so some of the plot groupings have been excluded, as they contain too few plots to give a meaningful indication of change. Such groupings are indicated with a '-' in table 2.

This analysis is able to detect the most subtle shifts in functional composition over the time period.

Crops ecotope group again shows changes in the arable landscape but not in the pastoral landscape. Changes suggest an increased disturbance regime, favouring large seeded, tall species adapted to frequent disturbance. The same change is indicated when all crop plots are looked at together, demonstrating the overwhelming effect of the large number of arable landscape plots over the pastoral landscape plots.

Tall grassland shows very little evidence of change in this analysis. The 4 correlations for arable landscape hedge plots did not provide any evidence of a consistent process of change. The results for all arable plots suggested that eutrophication may be occurring, with very reliable traits positively correlated. It could be that the changes detected in the 'simple' analysis were largely the result of shifts in the plots that altered enough to change ecotope group between 1978 and 1990. In the 'simple' analysis over 40% of the 1978 plots were classified differently in 1990 and 55% of the 1990 plots came from different ecotope groups in 1978. This could be an example of the change detected in the first analysis being largely due to change in a limited part of the landscape.

Eutrophic grassland again shows very little change other than in the arable landscape main plots. Here the correlations give quite strong evidence of eutrophication occurring. This may have been masked in the previous analysis by plots that changed between ecotope groups between 1978 and 1990. The 1990 group contained over 40% of plots from several other 1978 ecotope groups making consistent shifts difficult to detect given the variety of starting points.

Lowland grassland shows changes occurring in road verges in the pastoral and marginal upland landscapes, to streamsides in the pastoral landscape, and also when all of these plot types are analysed for all landscape types.

All of the groupings show consistent increases in large, long-lived species able to dominate the vegetation indicating eutrophication or dereliction or both. The characteristics of decreasing species do not help to decide between these possibilities, giving evidence of both. It is likely that both are occurring. These changes are in accordance with findings of the 'simple' analysis. Here the results from smaller plot groupings all indicate the same processes, a conclusion strengthened by detection of the same process at higher level groupings of the plots.

Lowland woods and hedges. Relatively few plots were available for analysis in the individual landscape types, however both arable and pastoral landscapes showed increases of large, competitive species. The results for all plot types in both landscapes tend further to indicate eutrophication. Stress tolerant species from species rich habitats are shown to be decreasing at the expense of the previously mentioned species. This result is further backed up by the results of analysis of all ecotope group 5 plots. These correlations strongly suggest eutrophication in these plots.

Acid woods showed no evidence of functional shifts between 1978 and 1990. Comparing this with the results of the 'simple' analysis, it may be that the very distinct shifts seen there were due to changes in those plots that changed enough to shift to another ecotope groups by 1990, other plots staying quite constant.

Upland grassland shows very little functional changes. Large numbers of correlations in the

marginal uplands and higher level groups are collections of distribution related traits not indicative of any consistent process of change.

Bogs and heaths. No change.

Summary.

‘Stay the same’ analysis. *Detecting subtle changes in plots whose vegetation remained superficially the same.*

Increased disturbance to **crops** group especially in the arable landscape.

Possible eutrophication of **tall grassland** in arable landscape.

Eutrophication of **eutrophic grassland** main plots in arable landscape.

Eutrophication and/or dereliction in **lowland grassland** road verges and streamsides.

Eutrophication of **lowland woods and hedges** in the pastoral landscape.

‘78-Based’ Analysis

This analysis follows the fate of plots, many of which may have changed ecotope groups between 1978 and 1990 i.e. plots that have been subject to fairly extreme changes in management. The matrices of change for plots in different landscape types can therefore act as a verification for some of the conclusions reached in the functional analyses.

Crops show quite distinct patterns of change across landscape types. Changes in many groups all point to the same conclusion. In arable and pastoral landscapes species of arable habitats with long-lived seed banks are decreasing at the expense of longer-lived, larger species characteristic of various grassland and derelict habitats. This suggests dereliction and is in accordance with the change of 30% of crop plots to various grassland types.

Tall grassland shows more changes. Hedgerow plots show fairly consistent indications of change across landscape types and when taken as a whole. Ruderal species of arable and regularly managed habitats are giving way to large-seeded, large competitive species

characteristic of shady and wooded habitats. This indicates dereliction of these plots. Many plots accordingly moved into the lowland woods and hedges ecotope group over the survey period. As matrices of change are not available for individual plot types, this cannot be reliably verified. In other plot types, no changes were detected.

Eutrophic grassland shows hints of the same processes occurring in road verges in both the arable and pastoral landscapes and in arable streamside plots, and also in all pastoral plot types together. All these groups show an increase in large, long-lived, competitive species at the expense of ruderal species i.e. dereliction. The lack of more definitive indications of processes of change may be due to the variety of directions in which plots moved between 1978 and 1990. In both arable and pastoral landscapes huge numbers of plots moved from eutrophic grassland to other grassland ecotope groups and also to the crops group. These changes are very different functionally and would give a very confused picture.

The dereliction may be due to the 21% shift of plots from eutrophic grassland to tall grassland.

Lowland grassland shows fairly consistent indications of dereliction and in some cases eutrophication across the pastoral landscape and throughout road verge plots, these trends also show up in higher summary groups of plots.

Throughout road verges the trend is towards large, long-lived competitive species at the expense of smaller, short-lived species i.e. dereliction. There is also a hint in the less intensive marginal upland landscape of eutrophication accompanying dereliction, as stress tolerant species of species rich habitats are also decreasing.

Dereliction is suggested in pastoral streamsides and all streamsides together giving a suggestion of dereliction of lowland grassland throughout the pastoral landscape.

As a whole, the ecotope group shows signs of eutrophication and dereliction which could be linked to the large shifts of lowland grassland plots to tall grassland and eutrophic grasslands (more derelict and eutrophic types) as well as to woodland types.

Lowland woods and hedges show changes in both arable and pastoral landscapes and in the ecotope group throughout all landscape types. For the 2 landscape groups, the sets of

correlations are fairly consistent and show changes occurring as woodland species and species of shady habitats are lost to be replaced by species of more managed habitats. Several correlations with the 'hard' nutrient concentrations traits suggest increases in species of more nutrient rich habitats i.e. eutrophication. This is in agreement with observed shifts of over 25% of plots from group 5 to the more intensively managed tall grassland group between 1978 and 1990.

Acid woods show little change. Numbers of plots in individual landscapes are quite low and as a whole, plots moved from group 6 to the less intensive group 7 and to the more intensively managed lowland grassland and woodland groups, giving little net consistent shift.

The only evidence for a consistent process of change is in streamside plots where loss of stress tolerant species of woodland and shady places is at the expense of short-lived species of more managed habitats suggesting at least increases in disturbance of these plots.

Upland grassland shows no overwhelming evidence of change. The correlations are rather a mixture, possibly in accordance with the mixture of shifts in plots between 1978 and 1990.

Plots moved to bogs/heaths, woodland and grassland groups.

Bogs / heaths show a mixture of correlations within each grouping of plots that as a group would be rather inconclusive. However, the pattern of correlations is very consistent across all groupings so some interpretation is possible. All show losses of large-seeded stress tolerant species at the expense of ruderal species of arable and regularly cut habitats with high SLAs. This very consistent shift would be consistent with eutrophication and increased disturbance (more intense management) and may coincide with shifts to upland grassland.

Summary.

'78-based' analysis. *Following the divergent fate of plots from a common starting point.*

Dereliction of **crops** group in arable and pastoral landscapes.

Dereliction of **tall grassland** hedgerows in lowland landscape types.

Dereliction of **eutrophic grassland** in the pastoral landscape and in road verges.

Dereliction of **eutrophic grassland** streamside plots in the arable landscape.

Dereliction of **lowland grassland** in the pastoral landscape and in hedgerows and streamsides.

Eutrophication and increased disturbance in **lowland woods and hedges**.

Increased disturbance in **acid woodland** streamside plots.

Eutrophication and increased disturbance in **bogs and heaths**.

Table 1. Simple analysis.

Functional changes in ecotope groups between 1978 and 1990. Data for each year grouped according to the ecotope group in which the plot was classified in that year. Eu - correlations between species proportional changes and values of species traits consistently indicate eutrophication in those vegetation plots. De - the correlations consistently indicate dereliction of the vegetation in the plots. Di - correlations consistently indicate increased disturbance to the vegetation in those plots. np - correlations indicate no consistent process of change occurring in the vegetation detectable in the species traits used. Asterisks indicate plot groupings that do not occur or contain too few plots.

Landscape type	Ecotope group	Plot type				
		Hedges	Road verges	Streamsid es	Main plots	All plots
Arable	Crops (1)	*	*	*	Di	Di
	Tall grassland (2)	De	De	np	*	np
	Eutrophic g'land (3)	*	np	np	np	np
	Lowland g'land (4)	*	*	Di	Di	Di
	Lowland wds/hdgs (5)	Eu	*	Eu	*	Eu
	Acid woods (6)	*	*	*	*	*
	Upland g'land (7)	*	*	*	*	*
	Bogs/heaths (8)	*	*	*	*	*
Marginal upland	Crops (1)	*	*	*	*	*
	Tall grassland (2)	np	*	*	*	*
	Eutrophic g'land (3)	*	np	*	np	np
	Lowland g'land (4)	np	np	Eu	Eu	np
	Lowland wds/hdgs (5)	*	*	*	*	*
	Acid woods (6)	*	*	np	*	np
	Upland g'land (7)	*	*	np	np	np
	Bogs/heaths (8)	*	*	np	np	np
Pastural	Crops (1)	*	*	*	np	np
	Tall grassland (2)	np	np	np	*	np
	Eutrophic g'land (3)	*	np	np	np	np
	Lowland g'land (4)	np	De/Eu	De/Eu	np	Eu
	Lowland wds/hdgs (5)	np	*	np	*	np
	Acid woods (6)	*	*	np	*	*
	Upland g'land (7)	*	*	*	*	*
	Bogs/heaths (8)	*	*	*	*	*
Upland	Crops (1)	*	*	*	*	*
	Tall grassland (2)	*	*	*	*	*
	Eutrophic g'land (3)	*	*	*	*	*
	Lowland g'land (4)	*	*	*	*	*
	Lowland wds/hdgs (5)	*	*	*	*	*
	Acid woods (6)	*	*	*	Eu	De/Eu
	Upland g'land (7)	*	*	np	np	np
	Bogs/heaths (8)	*	*	np	np	np

All landscape types	Plot Type				
Ecotope Group	Hedges	Road verges	Streamsid es	Main plots	All plots
Crops (1)	*	*	*	np	np
Tall grassland (2)	np	np	np	*	np
Eutrophic g'land (3)	*	np	*	np	np
Lowland g'land (4)	np	np	np	np	np
Lowland wds/hdgs (5)	Eu	*	*	*	Eu
Acid woods (6)	*	*	np	*	De
Upland g'land (7)	*	*	np	np	np
Bogs/heaths (8)	*	*	*	np	np

Table 2. Staysame analysis.

Functional changes in ecotope groups between 1978 and 1990. Data from plots that were classified as belonging to the same ecotope group in 1978 and 1990. Eu - correlations between species proportional changes and values of species traits consistently indicate eutrophication in those vegetation plots. De - the correlations consistently indicate dereliction of the vegetation in the plots. Di - correlations consistently indicate increased disturbance to the vegetation in those plots. np - correlations indicate no consistent process of change occurring in the vegetation detectable in the species traits used. Asterisks indicate plot groupings that do not occur or contain too few plots. Hyphens are plot groupings that contain too few plots for this particular analysis.

Landscape type	Ecotope group	Plot type				
		Hedges	Road verges	Streamsides	Main plots	All plots
Arable	Crops (1)	*	*	*	Di	Di
	Tall grassland (2)	np	np	np	*	Eu
	Eutrophic g'land (3)	*	np	-	Eu	Eu
	Lowland g'land (4)	*	*	np	np	np
	Lowland wds/hdgs (5)	Eu	*	-	*	Eu/De
	Acid woods (6)	*	*	*	*	*
	Upland g'land (7)	*	*	*	*	*
	Bogs/heaths (8)	*	*	*	*	*
Marginal upland	Crops (1)	*	*	*	*	*
	Tall grassland (2)	-	*	*	*	*
	Eutrophic g'land (3)	*	-	*	-	np
	Lowland g'land (4)	-	De/Eu	-	np	np
	Lowland wds/hdgs (5)	*	*	*	*	*
	Acid woods (6)	*	*	np	*	np
	Upland g'land (7)	*	*	np	np	np
	Bogs/heaths (8)	*	*	-	np	np
Pastural	Crops (1)	*	*	*	np	np
	Tall grassland (2)	np	np	-	*	np
	Eutrophic g'land (3)	*	np	-	np	-
	Lowland g'land (4)	-	De/Eu	De/Eu	np	np
	Lowland wds/hdgs (5)	np	*	-	*	Eu
	Acid woods (6)	*	*	np	*	*
	Upland g'land (7)	*	*	*	*	*
	Bogs/heaths (8)	*	*	*	*	*
Upland	Crops (1)	*	*	*	*	*
	Tall grassland (2)	*	*	*	*	*
	Eutrophic g'land (3)	*	*	*	*	*
	Lowland g'land (4)	*	*	*	*	*
	Lowland wds/hdgs (5)	*	*	*	*	*
	Acid woods (6)	*	*	*	np	np
	Upland g'land (7)	*	*	np	np	np
	Bogs/heaths (8)	*	*	np	np	np

All landscape types	Plot Type				
Ecotope Group	Hedges	Road verges	Streamsides	Main plots	All plots
Crops (1)	*	*	*	np	Di
Tall grassland (2)	np	np	np	*	np
Eutrophic g'land (3)	*	np	*	np	np
Lowland g'land (4)	np	De/Eu	De/Eu	np	np
Lowland wds/hdgs (5)	np	*	*	*	Eu
Acid woods (6)	*	*	np	*	np
Upland g'land (7)	*	*	np	np	np
Bogs/heaths (8)	*	*	*	np	np

Table 3. 78 based analysis.

Functional changes in ecotope groups between 1978 and 1990. Data for each year grouped according to the ecotope group in which the plot was classified in 1978. Eu - correlations between species proportional changes and values of species traits consistently indicate eutrophication in those vegetation plots. De - the correlations consistently indicate dereliction of the vegetation in the plots. Di - correlations consistently indicate increased disturbance to the vegetation in those plots. np - correlations indicate no consistent process of change occurring in the vegetation detectable in the species traits used. Asterisks indicate plot groupings that do not occur or contain too few plots.

Landscape type	Ecotope group	Plot type				All plots
		Hedges	Road verges	Streamsid es	Main plots	
Arable	Crops (1)	*	*	*	np	De
	Tall grassland (2)	De	np	np	*	np
	Eutrophic g'land (3)	*	De	De	np	np
	Lowland g'land (4)	*	*	np	De	np
	Lowland wds/hdgs (5)	Eu	*	np	*	Eu
	Acid woods (6)	*	*	*	*	*
	Upland g'land (7)	*	*	*	*	*
	Bogs/heaths (8)	*	*	*	*	*
Marginal upland	Crops (1)	*	*	*	*	*
	Tall grassland (2)	np	*	*	*	*
	Eutrophic g'land (3)	*	np	*	np	np
	Lowland g'land (4)	np	De/Eu	np	np	np
	Lowland wds/hdgs (5)	*	*	*	*	*
	Acid woods (6)	*	*	np	*	np
	Upland g'land (7)	*	*	np	np	np
	Bogs/heaths (8)	*	*	np	np	np
Pastural	Crops (1)	*	*	*	De	De
	Tall grassland (2)	De	np	np	*	np
	Eutrophic g'land (3)	*	De	np	np	De
	Lowland g'land (4)	np	De	De	np	De
	Lowland wds/hdgs (5)	np	*	np	*	Eu
	Acid woods (6)	*	*	np	*	*
	Upland g'land (7)	*	*	*	*	*
	Bogs/heaths (8)	*	*	*	*	*
Upland	Crops (1)	*	*	*	*	*
	Tall grassland (2)	*	*	*	*	*
	Eutrophic g'land (3)	*	*	*	*	*
	Lowland g'land (4)	*	*	*	*	*
	Lowland wds/hdgs (5)	*	*	*	*	*
	Acid woods (6)	*	*	*	np	np
	Upland g'land (7)	*	*	np	np	np
	Bogs/heaths (8)	*	*	Eu/Di	np	Eu/Di

All landscape types	Plot Type				All plots
Ecotope Group	Hedges	Road verges	Streamsid es	Main plots	
Crops (1)	*	*	*	De	De
Tall grassland (2)	De	np	np	*	np
Eutrophic g'land (3)	*	np	*	np	np
Lowland g'land (4)	np	De	De	np	De
Lowland wds/hdgs (5)	np	*	*	*	Eu
Acid woods (6)	*	*	Di	*	np
Upland g'land (7)	*	*	np	np	np
Bogs/heaths (8)	*	*	*	Eu/Di	Eu/Di
Grand Total					

Table 1. Simple analysis.

Functional changes in ecotope groups between 1978 and 1990.

Data for each year grouped according to the ecotope group in which the plot was classified in that year.

Figures are number of significant correlations ($P=0.05$) between species proportional change and values of species traits. Figures in bold are correlations that indicate consistent functional changes to the vegetation.

Figures in parentheses indicate correlations of uncertain ecological meaning. Asterisks indicate plot groupings that do not occur or contain too few plots.

Landscape type	Ecotope group	Plot type				
		Hedges	Road verges	Streamsid es	Main plots	All plots
Arable	Crops (1)	*	*	*	2	4
	Tall grassland (2)	8	5	(2)	*	(4)
	Eutrophic g'land (3)	*	(0)	(1)	(0)	(0)
	Lowland g'land (4)	*	*	4	8	6
	Lowland wds/hdgs (5)	2	*	4	*	4
	Acid woods (6)	*	*	*	*	*
	Upland g'land (7)	*	*	*	*	*
	Bogs/heaths (8)	*	*	*	*	*
Marginal upland	Crops (1)	*	*	*	*	*
	Tall grassland (2)	(1)	*	*	*	*
	Eutrophic g'land (3)	*	(4)	*	(4)	(5)
	Lowland g'land (4)	(4)	(2)	5	4	(5)
	Lowland wds/hdgs (5)	*	*	*	*	*
	Acid woods (6)	*	*	(2)	*	(0)
	Upland g'land (7)	*	*	(2)	(3)	(1)
	Bogs/heaths (8)	*	*	(6)	(1)	(0)
Pastural	Crops (1)	*	*	*	(1)	(1)
	Tall grassland (2)	(1)	(2)	(2)	*	(0)
	Eutrophic g'land (3)	*	(1)	(1)	(1)	(3)
	Lowland g'land (4)	(2)	6	3	(3)	7
	Lowland wds/hdgs (5)	(1)	*	(4)	*	(1)
	Acid woods (6)	*	*	(4)	*	*
	Upland g'land (7)	*	*	*	*	*
	Bogs/heaths (8)	*	*	*	*	*
Upland	Crops (1)	*	*	*	*	*
	Tall grassland (2)	*	*	*	*	*
	Eutrophic g'land (3)	*	*	*	*	*
	Lowland g'land (4)	*	*	*	*	*
	Lowland wds/hdgs (5)	*	*	*	*	*
	Acid woods (6)	*	*	*	3	10
	Upland g'land (7)	*	*	(1)	(4)	(4)
	Bogs/heaths (8)	*	*	(1)	(3)	(3)

All landscape types	Plot Type				
Ecotope Group	Hedges	Road verges	Streamsid es	Main plots	All plots
Crops (1)	*	*	*	(2)	(2)
Tall grassland (2)	(7)	(3)	(4)	*	(2)
Eutrophic g'land (3)	*	(0)	*	(3)	(1)
Lowland g'land (4)	(3)	(6)	(4)	(6)	8
Lowland wds/hdgs (5)	2	*	*	*	4
Acid woods (6)	*	*	(7)	*	11
Upland g'land (7)	*	*	(0)	(4)	(6)
Bogs/heaths (8)	*	*	*	(5)	(3)

Table 2. Staysame analysis.

Functional changes in ecotope groups between 1978 and 1990.

Data from plots that were classified as belonging to the same ecotope group in 1978 and 1990.

Figures are number of significant correlations ($P=0.05$) between species proportional change and values of species traits. Figures in bold are correlations that indicate consistent functional changes to the vegetation.

Figures in parentheses indicate correlations of uncertain ecological meaning. Asterisks indicate plot groupings that do not occur or contain too few plots. Hyphens are plot groupings that contain too few plots for this particular analysis.

Landscape type	Ecotope group	Plot type				
		Hedges	Road verges	Streamsid es	Main plots	All plots
Arable	Crops (1)	*	*	*	8	8
	Tall grassland (2)	(4)	(0)	(0)	*	4
	Eutrophic g'land (3)	*	(1)	-	8	4
	Lowland g'land (4)	*	*	(1)	(0)	(1)
	Lowland wds/hdgs (5)	3	*	-	*	3
	Acid woods (6)	*	*	*	*	*
	Upland g'land (7)	*	*	*	*	*
	Bogs/heaths (8)	*	*	*	*	*
Marginal upland	Crops (1)	*	*	*	*	*
	Tall grassland (2)	-	*	*	*	*
	Eutrophic g'land (3)	*	-	*	-	(2)
	Lowland g'land (4)	-	4	-	(4)	(0)
	Lowland wds/hdgs (5)	*	*	*	*	*
	Acid woods (6)	*	*	(1)	*	(3)
	Upland g'land (7)	*	*	(1)	(5)	(5)
	Bogs/heaths (8)	*	*	-	(0)	(1)
Pastural	Crops (1)	*	*	*	(0)	(0)
	Tall grassland (2)	(2)	(2)	-	*	(2)
	Eutrophic g'land (3)	*	(0)	-	(0)	-
	Lowland g'land (4)	-	7	7	(0)	(2)
	Lowland wds/hdgs (5)	(1)	*	-	*	4
	Acid woods (6)	*	*	(1)	*	*
	Upland g'land (7)	*	*	*	*	*
	Bogs/heaths (8)	*	*	*	*	*
Upland	Crops (1)	*	*	*	*	*
	Tall grassland (2)	*	*	*	*	*
	Eutrophic g'land (3)	*	*	*	*	*
	Lowland g'land (4)	*	*	*	*	*
	Lowland wds/hdgs (5)	*	*	*	*	*
	Acid woods (6)	*	*	*	(0)	(0)
	Upland g'land (7)	*	*	(2)	(0)	(1)
	Bogs/heaths (8)	*	*	(6)	(2)	(2)

All landscape types	Plot Type				
Ecotope Group	Hedges	Road verges	Streamsid es	Main plots	All plots
Crops (1)	*	*	*	(2)	5
Tall grassland (2)	(2)	(2)	(1)	*	(1)
Eutrophic g'land (3)	*	(4)	*	(2)	(2)
Lowland g'land (4)	(3)	5	4	(0)	(2)
Lowland wds/hdgs (5)	(4)	*	*	*	6
Acid woods (6)	*	*	(1)	*	(2)
Upland g'land (7)	*	*	(1)	(5)	(6)
Bogs/heaths (8)	*	*	*	(2)	(2)

Table 3. 78 based analysis.

Functional changes in ecotope groups between 1978 and 1990.

Data for each year grouped according to the ecotope group in which the plot was classified in 1978.

Figures are number of significant correlations ($P=0.05$) between species proportional change and values of species traits. Figures in bold are correlations that indicate consistent functional changes to the vegetation.

Figures in parentheses indicate correlations of uncertain ecological meaning. Asterisks indicate plot groupings that do not occur or contain too few plots.

Landscape type	Ecotope group	Plot type				All plots
		Hedges	Road verges	Streamsidess	Main plots	
Arable	Crops (1)	*	*	*	(3)	8
	Tall grassland (2)	8	(4)	(4)	*	(4)
	Eutrophic g'land (3)	*	7	2	(3)	(4)
	Lowland g'land (4)	*	*	(0)	5	(6)
	Lowland wds/hdgs (5)	2	*	(0)	*	6
	Acid woods (6)	*	*	*	*	*
	Upland g'land (7)	*	*	*	*	*
	Bogs/heaths (8)	*	*	*	*	*
Marginal upland	Crops (1)	*	*	*	*	*
	Tall grassland (2)	(0)	*	*	*	*
	Eutrophic g'land (3)	*	(3)	*	(0)	(4)
	Lowland g'land (4)	(1)	12	(4)	(1)	(5)
	Lowland wds/hdgs (5)	*	*	*	*	*
	Acid woods (6)	*	*	(0)	*	(5)
	Upland g'land (7)	*	*	(2)	(1)	(2)
	Bogs/heaths (8)	*	*	(2)	(1)	(0)
Pastural	Crops (1)	*	*	*	8	9
	Tall grassland (2)	7	(2)	(0)	*	(0)
	Eutrophic g'land (3)	*	4	(2)	(1)	6
	Lowland g'land (4)	(3)	6	10	(3)	12
	Lowland wds/hdgs (5)	(1)	*	(4)	*	10
	Acid woods (6)	*	*	(5)	*	*
	Upland g'land (7)	*	*	*	*	*
	Bogs/heaths (8)	*	*	*	*	*
Upland	Crops (1)	*	*	*	*	*
	Tall grassland (2)	*	*	*	*	*
	Eutrophic g'land (3)	*	*	*	*	*
	Lowland g'land (4)	*	*	*	*	*
	Lowland wds/hdgs (5)	*	*	*	*	*
	Acid woods (6)	*	*	*	(3)	(6)
	Upland g'land (7)	*	*	(6)	(3)	(2)
	Bogs/heaths (8)	*	*	5	(3)	8

All landscape types	Plot Type				All plots
Ecotope Group	Hedges	Road verges	Streamsidess	Main plots	
Crops (1)	*	*	*	10	12
Tall grassland (2)	9	(6)	(4)	*	(3)
Eutrophic g'land (3)	*	(5)	*	(1)	(4)
Lowland g'land (4)	(4)	12	10	(2)	11
Lowland wds/hdgs (5)	(1)	*	*	*	11
Acid woods (6)	*	*	9	*	(5)
Upland g'land (7)	*	*	(3)	(2)	(5)
Bogs/heaths (8)	*	*	*	6	8
Grand Total					

Review of farm management practices

The attached tables summarise the principle changes in management practice that effect botanical diversity. These have been drawn up in part from the ADAS farm management review and in part from consultation with BTO on the farm birds project. As will be seen the information in the lowlands is far more reliable than the uplands. Many farm management practices change concurrently and it is not possible therefore to disaggregate their effects.

Changes in management practices between 1970 and 1990 (based on available figures)

Aggregate class I: Crops/weeds	
Fertiliser	+
Post-emergent herbicides	+
Pre-emergent herbicides	x3
Insecticides	x2
Fungicides	*
Molluscicides	*
Seed dressings	+
Autumn sowing	*
Aggregate Class III/IV (part): Fertile grasslands/variable grasslands	
Fertiliser	=
Slurry	+
Herbicides	+
Yield	+
Cattle	=
Sheep	+
= no change + increase by >20% x2 increase by >200% x3 increase by >300% * new practice - introduced since 1970	

Management practices likely to have increased between 1970 and 1990

Aggregate class II: Tall grass/herbs

Flailing

Aggregate class V: Lowland wooded

Flailing (hedge)

Dutch elm disease (wood)

Aggregate class VI: Upland wooded

Canopy increase

Aggregate class VII: Moorland

Afforestation

Grazing patterns

N deposition

Aggregate class VIII: Heathland/bog

Afforestation

Grazing patterns

N deposition

Evaluating change in the distribution of common species between 1978 and 1990 by means of logistic regression.

Introduction

Several common grassland species showed a decline in abundance between the Countryside Surveys of 1978 and 1990 whilst other species increased. Different species may have declined for different reasons such as climate and/or soil conditions. For example, *Polygonum aviculare*, *Bellis perennis*, *Lotus corniculatus*, *Centaurea nigra* and *Succisa pratensis* may have decreased and *Cirsium arvense*, *Urtica dioica* and *Galium aparine* may have increased because of increased productivity of grassland.

Before the development of the British "Ellenberg" values and their subsequent use in this project it was assumed the best method of determining the reasons for change in the vegetation in the plots from the Countryside surveys of 1978 and 1990 would be through multiple regression techniques. The regression models would make use of national datasets on climate and soil. There was always perceived to be a problem of scale with climate values because the baseline climate data provided by the TIGER programme by the Climatic Research Unit (CRU) were interpolated to a 10-km scale whereas the Countryside Survey squares are on a 1-km scale. Soils data were considered to be on the same scale as the survey data but the availability of data was not known.

In this short paper I have analysed a sub-set of the Countryside Survey data, on the change in abundance of fourteen common species and what factors can be correlated with those changes.

Methodology

An ordinal value that represented the change in the number of "X-plots" in which fourteen common species (Table 1) were found was created. The ordinal value had a range from -5 to

results. The Multiple regression of climate data alone produced significant models for *Juncus bulbosus* and *Succisa pratensis* (Table 2).

Juncus bulbosus

Multiple regression suggested that the increase in the abundance of *Juncus bulbosus* has occurred where the winter temperatures are low and the summer temperatures are warm whilst the species has decreased in areas where the winter is warm and the summer temperatures are relatively cool (Table 2).

Agrostis capillaris

Multiple regression showed no significant model. The Ordinal Logistic Regression, however, suggested that the significance in the model was in areas where this species has declined most and these areas are where it is cool in the winter and warm in the summer and the decline occurred over many different soil types (Table 2)

Succisa pratensis

The multiple regression model for this species showed that this species increased where the January temperature was warmest and decreased where it was coolest. The Ordinal Logistic Regression showed that the model was not significant to any change category (Table 2). The ordinal Logistic Regression also suggested that the species did best where the January temperature was coolest. No effects of different soil types were found.

Discussion

The modelling exercise used here has shown that the change in the abundance of three species can be attributed significantly to the climate in the areas in which they were found (Table 2).

However, our previous knowledge tells us that the results using climate variables are likely to be spurious.

The modelling exercise that was envisaged was to compare changing abundance with

+5 with a value of -5 showing maximum decline and +5 a maximum increase. A value of 0 showed no change. Plots where the species were found on neither occasion were removed from the analysis.

Climate data were assembled from the 10-km baseline data for Great Britain supplied by CRU. Three climate variables; mean January temperature, mean July temperature and annual precipitation, representing the thirty year mean from 1960-1990 were used as independent variables. In addition the percentage of sea in the surrounding 10-km square was used as a surrogate for oceanicity. 21 separate soil groups were identified from soil pits dug for the 1978 Countryside Survey. Each 1-km square had five pits dug (one for each X-plot). A means of determining the “mean” soil type for each 1-km square was required for this analysis. There were 85 different combinations of the 21 soil types identified in a preliminary assessment of the soil data which was considered to large a number for significance to be detected. A second assessment reduced the number of groups to 17.

Ordinal logistic regression models were attempted for each species. The change in the abundance of the species being the dependent variable and the climate data being the independent variables. As the soil data were not continuous they were included as factors. Several models failed to converge and those that did, produced results that were difficult to interpret. As the ordinal scale was quite long (11 points) multiple regression models were created using Genralized Linear Modelling. The soil data added a large number of degrees of freedom and were removed. Multiple regression was carried out using the climate data alone as independent variables.

Results

The Ordinal Logistic regression produced significant models for *Agrostis capillaris*, and *Succisa pratensis* (Table 2). The Generalized Linear Modelling produced no significant

changing climate variables and soil parameters. This was not possible as we do not have reliable climate data to represent change between 1978 and 1990 as we do not have a baseline 10-km climatology for 1950-1980. Soil data for key variables such as pH, N, P and K are available but vary so much between quadrats that trying to produce a 1-km square average would be extremely problematical. In the analysis we have done, the soil type would not vary between 1978 and 1990. We will always have a problem that concerns degrees of freedom in models. The more variables we use the more quadrat data we will need to use. The lack of convergence in some of the Ordinal Logistic Regression occurred because the number of quadrats containing some species was not large enough to cope with the large number of independent variables involved.

The results show that three species have changed in abundance in some areas of the country in a different way to others and that changes in the abundance of *Agrostis capillaris* have varied depending on soil type. Changes have been independent of mean rainfall, suggesting that the three species with significant models are not likely to be affected by changes in mean precipitation, although like most species they are likely to be affected by drought. It is likely that significant responses to temperature are acting as correlates to regional changes in landuse and agricultural practice.

This modelling approach is not as successful as the method using the "Ellenberg" scores of vegetation described elsewhere in this report. That methodology is more reliable as it uses information on warmth, moisture, and nutrients that is collected from the quadrat itself and does not depend on interpolated values from a national dataset (temperature and rainfall) or expensive and extremely localised information (soil data).

Table 1: The species used for the multiple regression models.

<i>Polygonum aviculare</i>	Knotgrass
<i>Bellis perennis</i>	Daisy
<i>Lotus corniculatus</i>	Birds Foot Trefoil
<i>Centaurea nigra</i>	Knapweed
<i>Juncus bulbosus</i>	Bulbous Rush
<i>Agrostis capillaris</i>	Common Bent
<i>Succisa pratensis</i>	Devil's Bit Scabious
<i>Rubus fruticosus</i>	Bramble
<i>Urtica dioica</i>	Stinging Nettle
<i>Galium aparine</i>	Cleavers
<i>Molinia caerulea</i>	Purple Moor Grass
<i>Nardus stricta</i>	Mat-Grass
<i>Calluna vulgaris</i>	Ling
<i>Cirsium arvense</i>	Creeping Thistle

Table 2: Regression models for *Juncus bulbosus*, *Agrostis capillaris*, and *Succisa pratensis*. Correlations for soil type are expressed as deviance from values for podzols.

<i>Juncus bulbosus</i>		Multiple Regression		
		coefficient	T	P
	constant	-4.46	-1.12	0.272
	Jan Temp	-0.72	-3.05	0.004 **
	July Temp	0.47	1.96	0.058
	Precipitation	0.00	-0.17	0.866
	% Sea	0.00	-0.73	0.471
	R2			21.5
<i>Agrostis capillaris</i>		Ordinal Logistic Regression		
		Coefficient	Z	P
	Brown Earth	-0.810	-0.8	0.424
	Gley	-2.414	-2.34	0.019 *
	Rendzina	-1.766	-1.09	0.275
	Calcareous	-4.463	-2.13	0.033 *
	Peat	-1.759	-1.56	0.120
	Peaty-Gley	-1.756	-1.12	0.265
	Podzol/Brown Earth	-1.089	-0.88	0.380
	Podzols & Peat	-1.985	-1.27	0.206
	Gley/Brown Earth, Gley & Brown Earth	-2.836	-2.70	0.007 **
	Brown Earth & Gley	-3.301	-2.65	0.008 **
	Brown Earth & Calcareous	-0.391	-0.25	0.802
	Peat & Peaty-Gley	-3.247	-2.70	0.007 **
	Mix of all major soil types	-2.059	-1.87	0.061
	Mix of Podzols & Peat	-3.102	-2.82	0.005 **
	Mix of Podzols, Rendzinas & Peat	-0.114	-0.05	0.957
	Mix of Brown Earth & Rendzinas	1.115	0.72	0.472
	Mix of Brown Earth, Rendzinas & Peat	2.073	0.98	0.327
	Mix of Podzols & Brown Earth	-2.456	-2.21	0.027 *
	Mix of Podzols, Brown Earth & Peat	-1.764	-1.44	0.150
	Mix of Brown Earth & Peat	-2.205	-2.15	0.031
	January Temperature	-0.606	-3.44	0.001 ***
	July Temperature	0.4441	2.29	0.022 *
	Precipitation	0.000	-0.40	0.690
	% Sea	0.008	1.57	0.117
	G = 43.87, df = 24			0.008 **

<i>Succisa pratensis</i>	Multiple Regression		
	Coefficient	T	P
constant	4.19	1.11	0.272
January Temp	1.07	4.64	0.001 ***
July Temp	-0.50	-1.82	0.075
Precipitation	0.00	-0.39	0.696
% Sea	-0.01	-1.25	0.220
R2			31.6

<i>Succisa pratensis</i>	Ordinal Logistic Regression		
	Coefficient	Z	P
Brown Earth	3.43	1.46	0.145
Gley	1.74	0.78	0.434
Rendzina	-2.16	-0.76	0.450
Peat	0.25	0.12	0.904
Peaty-Gley	2.59	0.95	0.341
Podzol/Brown Earth	3.03	1.08	0.278
Gley/Brown Earth, Gley & Brown Earth	1.05	0.49	0.624
Brown Earth & Calcareous	3.81	1.30	0.192
Peat & Peaty-Gley	-0.74	-0.34	0.731
Mix of all major soil types	0.63	0.29	0.78
Mix of Podzols & Peat	-0.45	-0.23	0.819
Mix of Brown Earth, Rendzinas & Peat	-1.96	0.71	0.475
Mix of Podzols, Brown Earth & Peat	-0.91	-0.34	0.732
Mix of Brown Earth & Peat	0.40	0.19	0.850
January Temperature	-1.49	-4.66	0.001 ***
July Temperature	0.47	1.08	0.282
Precipitation	0.00	0.93	0.353
% Sea	0.02	1.46	0.144
G = 39.51, df = 18			0.002 **